ISSN 0710-6815
ISBN 0-7727-4410-6

RISK PERSPECTIVES ON ENVIRONMENTAL IMPACT ASSESSMENT

Editors: A.P. Grima, C.D. Fowle and R.E. Munn

General Editor: D.M. Whelpdale

Environmental Monograph No. 9

Institute for Environmental Studies,
University of Toronto,
Toronto, Canada
M5S 1A4

August 1989

Pub. No. EM-9

ISSN 0710-6815
ISBN 0-7727-4410-6

ENVIRONMENTAL MONOGRAPH SERIES

(The works in this series have been refereed)

EM-1 Vulnerability, resilience and the collapse of society, a review of models and possible climatic applications
Peter Timmerman, April 1981, 42 pp.

EM-2 Urban natural areas: ecology and preservation
Editors W.A. Andrews and J.L. Cranmer-Bying, July 1981, 215 pp.

EM-3 Living with risk: environmental risk management in Canada
Editors I. Burton, C.D. Fowle and R.S. McCullough
September 1982, 247 pp.

EM-4 The detection and trends in wet deposition data: a report of a workshop
Rapporteur R.E. Munn, April 1984, 47 pp.

EM-5 Environmental impact assessment: the Canadian experience
Editors J.B.R. Whitney and V.W. Maclaren
February 1985, 197 pp.

EM-6 The identification of early indicators of CO_2 climate warming in Canada
R.E. Munn, June 1984, 52 pp.

EM-7 Climate impact assessment in the Great Lakes Basin: overview and research recommendations **(Out of print)**
P. Timmerman and A.P. Grima, November 1985, 32 pp.

EM-8 Information needs for risk management
C.D. Fowle, A.P. Grima and R.E. Munn, November 1988, 318 pp.

TABLE OF CONTENTS

PREFACE AND ACKNOWLEDGEMENTS

Risk characterization and evaluation is playing an increasing role in the Environmental Impact Assessment (EIA) of development projects. Earlier EIAs either ignored uncertainties or implicitly assumed that information about impacts and mitigation was known with near certainty. In a risk context, uncertainty about both impacts and the efficacy of mitigation should be explicitly recognized at the outset.

In the mid-1980s, several agencies came to recognize that the increasing use of risk principles in environmental assessment raised many significant and interesting issues. The Canadian Environmental Assessment Research Council (CEARC) made risk analysis and the management of uncertainty one of its first research priorities and it soon became apparent that risk assessment overlapped other important themes such as mitigation and compensation.

CEARC, Environment Canada, SCOPE (Scientific Committee on Problems of the Environment), National Research Council of Canada (NRC), Ontario Hydro and Health and Welfare Canada sponsored a working group at the Insitute for Environmental Studies (IES), University of Toronto. Dr. R.E. Munn coordinated the group until he accepted the position of Leader, Environment Program at the International Institute for Applied Systems Analysis (IIASA) in Laxenburg, Austria in March 1985 when Dr. A.P. Lino Grima became coordinator of the Group.

In April 1985 about twenty-five practitioners and researchers attended the Workshop on Risk Management and EIA at Seneca College, King City, Ontario. The papers in this monograph were first outlined in that workshop. Their review and revision has involved many scientists and practitioners in Canada and other countries. These colleagues have encouraged us and supported us in many ways. We thank them all, particularly the reviewers for their stimulating critiques. Ms. Gail Rania provided the administrative support for the project and Miss J. Retel typed numerous drafts of this monograph and contributed her editorial experience. The editors owe much to their resourcefulness and dedication.

The financial support of the agencies in the IES Working Group on Risk Assessment is very much appreciated. Their representatives on our Steering Committee included: P.J. Paradine (CIDA), M.H. Sadar (CEARC), R.J. Malvern (Ontario Hydro), D. Krewski (National Health and Welfare), Carol Miller and P. Rodgers (Environment Canada), D. Miller (NRC and SCOPE) and J.E. Dooley (IES). It is with deep sorrow that we note the passing away of one of our colleagues, Dr. Don Miller. We owe him a special debt for his enthusiastic encouragement and support. These colleagues generously volunteered their experience, time and scientific expertise; their moral support and insightful comments were instrumental in obtaining financial support for the workshop and this publication.

CHAPTER 1
INTRODUCTION: RISK CONCEPTS IN ENVIRONMENTAL IMPACT ASSESSMENT

P. Timmerman
Institute for Environmental Studies

SETTING THE STAGE

This book examines the application of principles and insights from the growing field of risk assessment to the process of environmental impact assessment (EIA). There are two reasons to believe that the time is ripe for such an examination: first, risk assessment has achieved a level of sophistication and coherence that entitles it to serious consideration as a tool for environmental management; second, there is strong social and political pressure to grapple with the contentious issues surrounding "risk" and uncertainty in EIA.

Proper consideration of what "risk" means - and what taking it into consideration entails - requires a diverse array of techniques and expertise from different disciplines, ranging from engineering to political philosophy. This volume reflects that diversity. The authors have been guided both by the goal of incorporating risk assessment into the EIA process, particularly in Canada; and by a general understanding of what special requirements of the EIA process can best be served by this new approach.

In Chapter 2, Paradine explores the general procedural and historical background to the increasing use of risk assessment as part of EIA. He describes the ready acceptability of EIA to the use of risk techniques, and notes that much of what is considered EIA in Canada could also be considered as risk assessment under another name. Dickerson, from an American perspective, presents a different historical and procedural evolution in the increasing use of risk based on the impact of changing regulations under the National Environmental Policy Act (NEPA), but concludes that similar pressures for improved rigour in the qualitative assessment of hazards are taking place in both Canada and the U.S.

This improved rigour is the topic of Chapter 4, in which Dooley surveys the principles and techniques for scientific analysis of risk. These techniques form the basis for any serious use of risk assessment as a new and unique addition to the EIA toolbox. But the usefulness of risk assessment is not just limited to its analytic base: one of its potential benefits is its ability to relate different factors together under one conceptual "umbrella". In Chapter 5, Pushchak widens the scope of the discussion to show how risk evaluation can be used to integrate discussions of technical, economic, and social risks, as well as possible trade-offs and benefits. For such discussions to be meaningful, both the nature of the risks, and the context within which they are to be found, need to be clearly understood. Only in this way can negotiations and compensations for risks be handled rationally and equitably.

In Chapter 6, Timmerman describes some of the difficulties in relating technical risk analysis to value-laden social contexts, a theme which is taken up in Chapter 7 by Grima in his exploration of the "growth industry" of conflicts over risk. He examines the various actors in the risk debate, especially at the community level. Duinker, in Chapter 8, examines the debate at the level of the regulators, while Malvern and Paterson present the perspective of

the project proponent. In the concluding chapter, Grima and Fowle reflect on current and future directions.

Throughout this volume, the authors have been conscious that - in the words of Malvern and Paterson* - we are not dealing with "a magic wand that waves away conflict and uncertainty", nor will the incorporation of risk into EIA do the trick either. What risk assessment can do is force a crystallization of concerns, through its technical requirements or through its drive towards the integration of disparate topics within a common framework. This crystallization may permit the real issues at stake to be identified, considered and resolved - given the most appropriate overall EIA process.

The rest of this introduction sketches out a preliminary description of the elements of risk assessment; and while it assumes that the reader has some knowledge of EIA, it also presents EIA as a counterpart of risk assessment so as to provide the interested reader a familiar path through what has become a fascinating, but already vastly diverse, set of risk perspectives, concepts and techniques.

RISK ASSESSMENT

The justification for risk assessment as a management tool is somehow to cope in a more rational way with the conflicts which have come into being in recent years over the handling of various proposed - or already existing - projects of a complex technological nature. These conflicts are occasionally due to misapprehensions or misguided projects; but on many other occasions they can be seen as a function of a more fundamental "waning of consensus" (Ronge 1982; Timmerman 1984). Risk assessment proposes a managerial process in which technical risk analysis is embedded in a more broadly based assessment format, which can be, in turn, embedded in overall societal risk management. In this way, the process moves from **facts** to **values.**

On one level, this must be a good thing. An unbiased observer of the recent past - and the current turbulence with regard to risks and hazards - cannot help but notice that the assessment and management of risk has by and large been haphazard. This applies with equal force to individuals, businesses, and governments. At a deeper level, however, the confusions over risk and the intense conflicts generated by these confusions are likely symptomatic of ethical, moral, and political conflicts that a "more rational" management process will not necessarily be able to resolve.

What then is "risk assessment"? Although the field has been in existence only a short time, there have already been a number of classic papers and overall appraisals such as the original paper of Starr (1969), the works of Rowe (1977), Kates (1978), Burton and Whyte (1980), Burton et al (1982), Krewski and Birkwood (1987), and Rogers and Bates (1982). Because of this book's concentration on environmental hazards and impacts, the definitions supplied by Kates (1978) may provide a good beginning:

> **Environmental hazard** is the threat potential posed to man or nature by events originating in or transmitted by the natural or built environment. **Risk assessment** is an appraisal of both the kinds and degrees of threat posed by an environmental hazard.

---

*Names of aurthors, unless followed by year of publication, refer to papers in this volume.

Kates further suggests that risk assessment has three interrelated components: **risk identification, risk estimation,** and **risk evaluation.** Most of the researchers and supporters of risk assessment agree that one of the core elements of a risk assessment - an element that separates risk assessment from other kinds of impact assessment - is the second of these components, risk estimation. Whether labelled as risk estimation, or technical risk analysis, or some variation on this terminology, this component of a risk assessment process is an attempt to estimate scientifically, mathematically, statistically, or in some other rigorous fashion the probabilities and the consequences causally associated with potentially hazardous events. This, it must be emphasized, is to some degree an idealisation, and that is why it implies mathematical expression or the analysis of some system abstracted from real world conditions (see Dooley, Chapter 4); because as soon as one is actually engaged in many forms of risk estimation in the real world, a whole range of "mensuration problems" come into play. Rowe (1977), for example, distinguishes - even at this earliest stage in the process - between "descriptive uncertainty" and "measurement uncertainty": that is, between absence of information about the elements of a system that thereby precludes one from fully describing what the system "is" and how it works, and absence of information about the relevant measurements one should apply to the elements (or variables) of such a system.

This means, to be blunt, that a system may be so complex that no probabilities can be assigned with confidence to its functioning, and the tracking down of direct consequences may be just as difficult. The earlier step of risk identification may be based on a much less rigorous set of probability and consequence determinations, e.g. common sense or social norms. This is of particular concern with low-probability, high-consequence risks: society may have no history of dealing with some risks, and so will not identify them as risks until they actually manifest themselves in a major incident. Until the incident at Three Mile Island, for example, the lack of coherent design of control rooms was not identified as a risk: afterwards, it was widely acknowledged that one major contributor to the increased risk of an accident was the confused and unrelated instrument panels in front of the harassed decision-makers.

Shading off from risk estimation on one side, then, we may find risk identification occasionally as a function of the **perception** of risk; just as, on the other side, risk evaluation will also involve subjective factors in the evaluation of whether some risks are to be given a greater importance than others. In some of the literature, risk identification and risk estimation are both subsumed under the larger category of **risk analysis** or **risk determination** so as to keep the question of evaluation separate, under either the term risk evaluation or risk assessment. In this monograph, we use the term **risk assessment** interchangeably with **risk evaluation.** These terms cover a broad territory, including the value judgements and trade-offs in appraising alternative plans for mitigating risks or their consequences. The next stage, **risk management** includes the choice of a preferred course of action and its implementation. We consider risk decisions to be iterative through all stages; management considerations, as noted above, may influence which risks are selected for analysis, the resources made available for assessment, priorities for implementation, and so on. In this sense, risk management includes all previous stages (cf. Grima et al. 1986). We are, however, aware that risk management is often used to refer only to the final steps of choosing a mitigating strategy and implementating it (e.g. Krewski and Birkwood 1987) and this implies that management is qualitatively different and separate from the previous steps. As we noted elsewhere, "as long as we are reasonaby clear

about what we mean and the definitions are relatively free from ambiguity, we need not become involved in a semantic debate" (Grima et al. 1986).

Each of these elements can be broken down into several subcomponents: Chapter 4 (Dooley Fig. 1) presents a general model of risk analysis proceeding through the following stages: activity analysis, process analysis, exposure analysis, consequence analysis, and risk/benefit analysis. Chapters 5 (Pushchak) and 6 (Timmerman) articulate some of the elements - and the constraints - operating during the risk evaluation stage of any project. It should, however, be kept in mind that the elements or categories of operation overlap and blend into each other.

Burton and Whyte (1980), for instance, in speaking about techniques for risk estimation, refer to modelling, monitoring, surveying, screening and testing as parts of an "environmental risk system", in part because the selection of techniques depends upon the earlier risk identification; and, in other cases, the identification of risks depends upon the estimation techniques already in place. This brings up the complex issue of how risks are interpreted. A current example is provided by the litigation over asbestosis. Because of the long period of time that elapses between exposure to asbestos fibres and the manifestation of asbestosis, both the epidemiology and assigning of responsibility for damage can be problematic. In the American courts, substantial sums - into the billions of dollars - are being contested, dependent in many cases on whether insurance liability begins at the moment of exposure or at manifestation (since different insurers may have insured the same company over time). It is being argued that manifestation begins to occur at the microscopic level at the moment of exposure, and so the two are identical: yet - dangerous as asbestos is - not every exposure necessarily leads to asbestosis.

In Chapter 9 (Malvern and Paterson), this problem of interpreting risk is referred to as one in which uncertainty surrounds: (a) exposure processes; and (b) effect processes. As our analytic ability improves, and as our understanding of diseases such as cancer moves closer and closer to fundamental cellular and genetic processes, the clear distinction between exposure and effect dims.

The strong focus on human health risks derived from the environment as a pathway or carrier of toxins, is not the only reason for attempting to understand environmental impacts or risk. Environmental risk is multi-dimensional, and the more one attempts to incorporate the risk to the environment *per se* into an overall assessment, the more one comes up against the fact that not only is environmental risk identification and estimation complex; but the risks are very often hard to conceptualise, and, furthermore, our knowledge and experience of environmental risks are usually limited (Bowonder 1981).

These complexities and uncertainties paradoxically put even greater significance on the overall risk assessment and management processes, since one has to evaluate both whether the earlier steps in the process were appropriate, and also whether the risks are significant. It is the role of the overall process (and the assessors in that process) to ensure that appropriate appraisals are made, and to determine ultimately whether the expected risks are **acceptable.**

ACCEPTABLE RISK

The end result of any risk assessment is to discover whether a risk is acceptable.

> **Acceptable risk** is a risk whose probability of occurrence is so small, whose consequences are so slight, or whose benefits (perceived or real) are so great that a person, group, or society is willing to take that risk (Munn quoted in Grima et al. 1986).

Part of this definition includes the notion of **risk/benefit analysis** which, by analogy with cost/benefit analysis, tries to relate risks and benefits in such a way that some comparison or summation can be made. In many cases such relationships may be hard to make exactly, but there is little doubt that they must be taken into account in any risk assessment that aspires to come up with some way of determining collectively whether some specific risks are acceptable. Because the issue of what constitutes acceptable levels of risk is so controversial, it is imperative that the process by which one determines appropriate probabilities, consequences, and benefits must itself be acceptable to the parties involved. If this is not the case, assessment practitioners may find themselves involved in an infinite sceptical regress, where the parties to a conflict neither agree on the implications of the information presented, nor on the integrity of the information itself, etc., ad infinitum.

As Pushchak (Chapter 5) details, the central question in risk evaluation is "How much benefit do individuals require to accept voluntarily additional risks to their health or well-being?" The question of what constitutes "benefits" is even more unexplored than the question of risks, due both to the difficulty of identifying the appropriate or relevant benefits, and the more familiar issue of the equitable distribution of identified benefits. These are crucial elements in the creation of an acceptable array of risks and benefits to a potentially affected area or community (see also Grima, Chapter 7).

If we assume that during the risk identification and risk estimation phases of the risk assessment we have been able to provide acceptable quantification, we may be unable to do so with benefits (further assuming that a form of risk/benefit analysis is proposed). Direct benefits may be easily quantified, while the indirect benefits may be diffuse and difficult to bound for the purposes of rigorous calculation.

The question of equitable distribution is tied up with how what is to be distributed is to be measured. The advantage of risk/benefit analysis - as with its progenitor, cost/benefit analysis - is that it is designed to come up with a "bottom line"; and, in so doing, assumes that everything will be measured (or at least indicated) by one standard measuring system. This, of course, is usually dollars, which serve as surrogate for expressed preference or utility. The disadvantage of risk/benefit analysis is that it comes up against the problem of incommensurability in a particularly stark form.

Many risks are personal and endanger life or general health, and our society refuses to attach dollar signs to human life or health in making these sorts of calculations. There is no doubt that tradeoffs based on monetary considerations are made every day (e.g. we buy only a limited number of very expensive high-technology pieces of medical equipment), but we do not formally:

(a) subject individuals to involuntary risk of death for the good of others in peacetime;

(b) refuse to pay open-ended medical costs once someone has become a patient in a public hospital in a public health-care system; and

(c) compare people's right to life or health based on income.

For better or worse, this stems from the bedrock egalitarianism of our social philosophy, and we could sum up its tenets in the following **social principle of risk:**

> Everyone has an absolute right not to be subjected to an involuntary increase in personal risk for the benefit of someone else.

Variations on this principle have been called the "trump" theory of rights (Dworkin 1978) in a democracy, and their workings can be seen in the often controversial siting of hazardous waste facilities (see Chapter 6, Timmerman).

One difficulty is that this sort of absolute right very rarely becomes clearly expressed, and what we find instead are situations where the persons affected are likely to achieve some benefits, but that the bulk of the benefits will either belong to a larger group of others, or to one specific beneficiary (e.g. a corporation).

In such cases, the ability to compare risks and benefits may be essential to determining such alternatives as compensation to a potentially affected community (Pushchak, Chapter 5). The availability of such alternatives, which have only begun to be explored or tested, is in part dependent on the solution to some of the pressing quantitative measurement issues already referred to above.

One of the features of risk that contribute to this seemingly intractable problem is risk perception. As described in greater detail in Chapter 6 (Timmerman), **perceived risk** has been defined as an estimate of risk made qualitatively by a person or group in society, but this by no means exhausts its complexity. There are two main stances with regard to perceived risk: first, the critical stance, which argues that there is some objective risk estimate which is being misperceived by (in many cases) the semi-informed public; and second, the interpretative stance, which acknowledges that there is some misperception going on, but that there are also deeper matters at stake, concerning the contexts within which decisions about risk, attribution of responsibility, and the shaping of life patterns in general are being formed.

To take just one facet of this issue, the burden of having to commit oneself to an unknown, and possibly open-ended series of future risks - however insignificant - is felt by citizens of communities faced with the siting of hazardous waste facilities. Examples of mismanaged technology in the past haunt these sorts of proposals: citizens find themselves part of a growing scepticism about assurances that we are able to handle future uncertainties with premeditated care. Discussions about the specific risks involved with Project X very rapidly can turn into arguments about the insecurity people feel about living in a turbulent world, where they learn of new risks on a daily basis (cf. AIDS, pesticides on fruit and vegetables, etc.).

This reminds us that risk, and the perception of risk, are often functions of people's basic conceptions of life, the context within which they find themselves having to make decisions under conditions of uncertainty; in what Burton and Whyte (1980) refer to as an "Environmental Risk System". Now, while

we can use some process or system to come to terms with some aspects of this situation, we are in many cases talking about the priorities of whole ways of life. Can we even begin to come to terms with this type of risk decisions?

One option is to put forward a good set of criteria for acceptable risk, and tailor a decision process to achieve those criteria. Burton (1982) suggests that we should determine if certain risks are reasonable or unreasonable, unreasonable ones being those wherein:

1. those at risk are not fully informed of the risks and benefits;

2. an equitable distribution of risks and benefits (including compensation) is lacking;

3. a less risky alternative exists that yields equal or greater benefit;

4. its acceptance has been achieved without a full opportunity for participation in the decision process by those at risk;

5. the value judgements involved in risk/benefit assessment have been hidden from view or not clearly articulated.

In Chapter 7 of this volume, Grima asks us to consider risk assessment and environmental assessment as decision-making processes, rather than as planning exercises; and suggests that a rational basis for such decision-making would consider using conflict resolution as a way of reaching agreement on the reasonableness or unreasonableness of accepting risks as described in Burton's agenda and other similar agendas. This is because, as we have already noted, discussions of risk are inherently controversial; but also because scientific and other uncertainties tend to undermine confidence in the competence of the process, and this further exacerbates controversy over the uncertainty of possible outcomes.

A further point to be made in this context is that what some people consider reasonable and unreasonable criteria for risk assessment and management may differ substantially from criteria held by others. The differences between experts and the public have already been mentioned, but the extension of risk further and further into areas of public decision-making calls other actors into play, actors who bring essentially different "rationalities" to bear on the situation. In this context we can use "rationalities" to refer to different methods of relating means to possible ends. Miller (1985) has proposed a model of decision-making in the context of risk management, a model which constitutes an iterative process among three different rationalities for decision-making (Fig. 1): public/market choice; scientific assessment; and political inference. In each of these reasoning processes, the interaction of information, activities, and concerns contribute to the creation of an overall perception of the biosphere (physical and social).

As noted above, the public's perception of uncertainty may depend in part on their stance as "stakeholders", people with more than just failed calculations at stake. The scientist's professional interest in uncertainty as a focal point for improving our understanding and predictive capacities gives him or her a different perspective. The third group, the politicians and bureaucrats, are in the position of potentially mediating between these perspectives in the name of the greater public good. Theirs is both an inferential task, and also one that may lead to intervention. And it is here, where risk theory

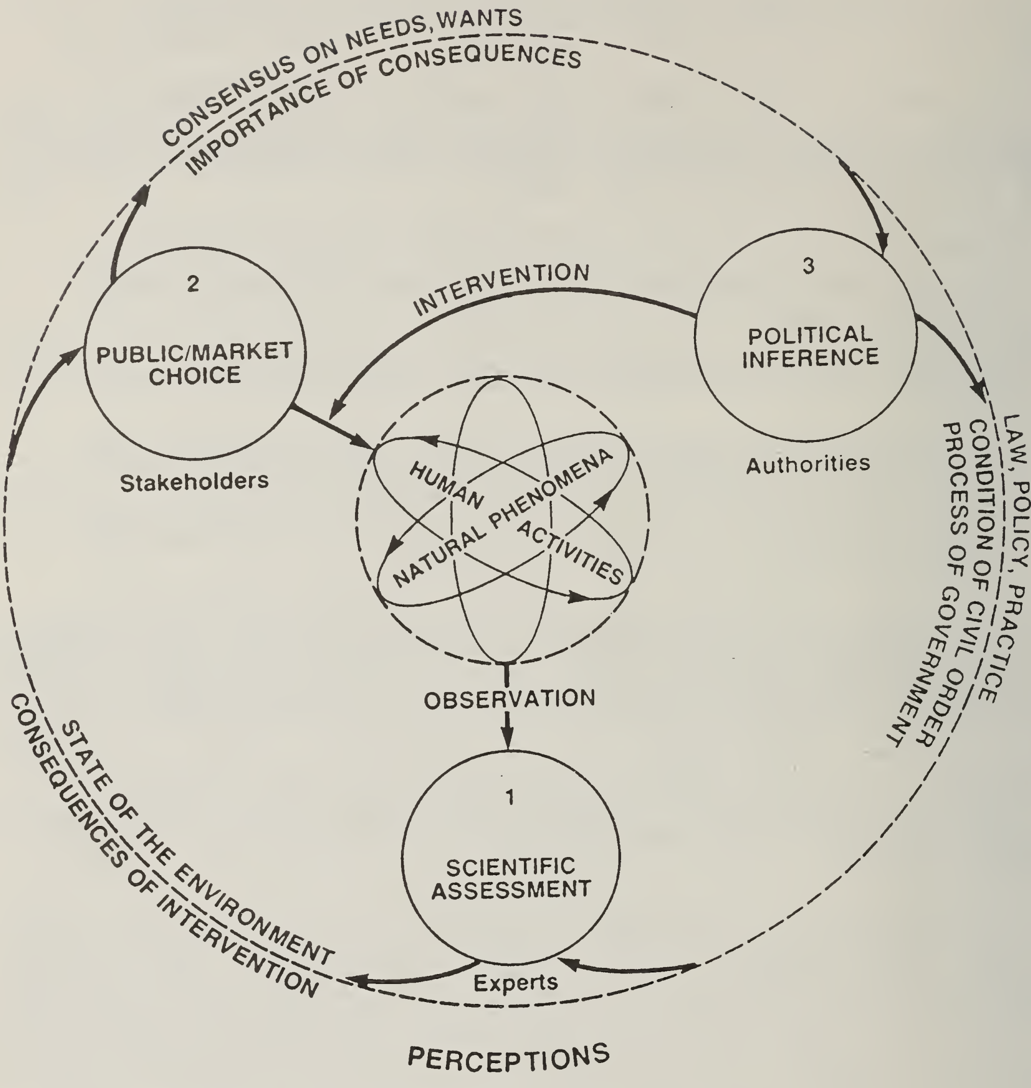

Fig. 1. Three rationalities: partners in decision-making
Source: Miller 1985

meets political practice, that the criteria for acceptability must finally be clarified.

In this context then, risk assessment is an attempt to bring the reduction of scientific uncertainty over probabilistic events together with the reduction of social uncertainty over personal and societal destinies, all focussed on the goal of "acceptable risk". One basic task is to reconcile the different approaches, methodologies, and vocabularies of risk (e.g. technical risk estimation versus perceived risk). There is also a need to expand the domain and range of risk as a concept so as to embrace more and more of the concerns which loom so large in the social and political arenas.

One method of approaching these tasks is to relate the special concerns of risk assessment to those other management processes that share similar or related concerns, while at the same time enhancing the concurrent development of other processes. This brings us to the stated objective of this volume, i.e. the application of risk approaches to environmental impact assessment particularly in Canada.

INCORPORATION OF RISK ASSESSMENT INTO ENVIRONMENTAL IMPACT ASSESSMENT

Environmental Impact Assessment was introduced into Canada at the federal level in 1973, when the federal Cabinet decided to create the Environmental Assessment and Review Process (EARP). Its objective was to take environmental impacts into account during the planning and implementation phases of federal projects. About the same time, EIA requirements were adopted in all of the provinces (Couch 1983), and since then the federal program has been considerably expanded.

The activities that the EARP process applies to include potentially hazardous activities such as offshore hydrocarbon development, nuclear energy projects, and transportation facilities such as pipelines and highways. Usually some form of risk analysis is carried out by the proponents and evaluators of such activities, whether it be economic risk analysis, or technical analysis of the possibility of mechanical failure in a component of a proposed system. As described in Chapter 2 (Paradine) of this volume, hydrocarbon developent has involved the greatest use of risk assessment principles to date on the federal scene, followed by the assessment of liquid natural gas transportation and storage, as well as the EIA for the Point Lepreau Nuclear Generating Station. These applications of risk concepts have been specific to each project: no mandated guidelines or procedures required for every EIA have been invoked - technical assessment of the risks is just one aspect of considering the potential impacts of a particular project.

In other jurisdictions the same situation recurs: while EIAs often consider risks as part of the assessment, the rationale for their consideration is underdeveloped, with the possible result that they are inconsistently handled, misconstrued, or their larger potential role in clarifying the goals and objectives of the EIA process may be obscured (Beanlands 1986).

One of the technical questions that has come to occupy a central position in various controversial EIAs is the treatment of low-probability, high-consequence (LOPHIC) risks. For a variety of reasons, particularly with respect to public hearings, this has become a very difficult problem to resolve. In the first place, the public perception of a risk often differs greatly from that obtained from engineering calculations; and, in any event, a layperson cannot usually distinguish between a probability of 10^{-8} and 10^{-6}. The public is much more concerned, by and large, with consequences (Timmerman, Chapter 6). The different priorities of technicians and the public led, for example, in the early days of EIA, to assessors underplaying or disregarding low probability risks in impact statements, even if the consequences might be serious. As elaborated in Chapter 2 (Paradine) and 3 (Dickerson), this situation is rapidly changing, and risk assessments of a broader type are looming larger and larger in both EIA panel considerations, and in the public sessions associated with the EIA process. Duinker (Chapter 8) envisages regulators, proponents, scientists and other stakeholders as major actors who collaborate to generate, review and legitimate the information base for an EIA, including risk assessment.

Paradine, Malvern and Paterson, Duinker, and other authors in this book, relate the concept of risk to EIA in the context of existing EIA practices. An idealisation of these practices, as put forward by Whitney and MacLaren (1985) is represented in Fig. 2. Duinker, for example, notes that many of the elements of the risk assessment match closely those of the EIA process; in a risk context, uncertainty about both the impacts and the efficacy of mitigations are explicitly recognized at the outset. In the idealised process, the overall notion is that of an "experiment" (Beanlands and Duinker 1983); that is, any new project should be considered as a perturbation of an existing situation, a perturbation that should confirm or refute a set of hypotheses (or predictions) about the resultant state of the environment after the experiment has begun. From this there follows **scoping** of the boundaries of the experiment, **prediction** of the probabilities and consequences, the **significance assessment** (i.e. what do these predictions entail, and does it matter?), and the **evaluation** (including the ultimate decision to proceed with a project). **Public participation** at most stages of the process is required; are after-construction **monitoring**, and **mitigation** measures.

RISK IN THE EIA SCHEMA

There is little doubt that risk figures throughout the whole EIA process, either formally or informally. To make this explicit, and to assist the reader in working through the possible applications described in the rest of this volume, let us make the following equivalences:

> During the **scoping** phase of EIA, risk identification is clearly important. Is there a history of risks associated with this particular technology? Are there risks we cannot identify now, and which will not appear in a standard EIA review, but which may develop in the future (e.g. new waste products)? Proponents often consider economic risks as crucial, but opponents of projects tend to underplay them. Are there ways of relating these risks to environmental risks in ways that allow the hearings to make appropriate risk/benefit trade-offs? In the initial screenings and setting up of preliminary guidelines for proponents, there are often implicit or explicit hazards identified that must be addressed. Are the risk identification procedures used at this stage adequate to minimize the discovery of new risks later on in the process?
>
> The **prediction** phase of the EIA process is one in which risk analyses have already been used frequently, and in which their future use is assured. Their obvious function is to improve the predictive capability by reducing the scientific uncertainty, and also by a more intense scoping of the boundaries of the potential impacts. The range of risks, their magnitude and scope, and their statistical probabilities of occurrence are where detailed risk identification and risk estimation could come into their own if data are available. Is it possible to use risk to smooth the process from scoping to prediction? Can risk analysis serve as the conceptual framework for the prediction phase of EIA, if one considers the predictive model to include everything from complete certainty to complete uncertainty, including those risks we can put probability distributions to?
>
> The **significance assessment** phase of EIA is usually associated with the "impact statement", the report giving the results of the overall assessment preparatory to review and hearings. The technical aspects of the subject are forbidding to all but the expert, and it is here, perhaps,

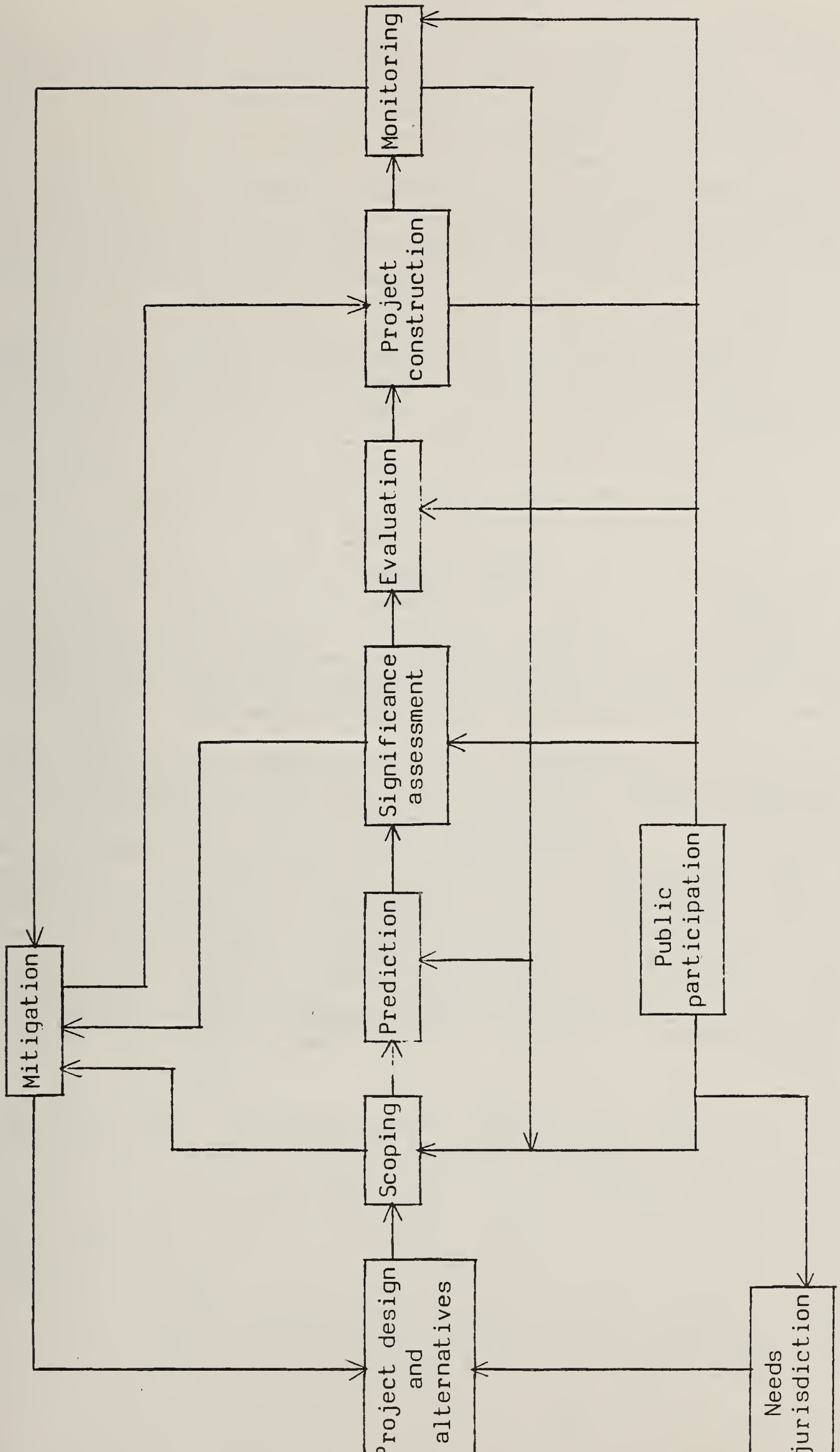

Fig. 2 Flow diagram of an ideal EIA process
Source: Whitney and Maclaren 1985

that the future usefulness of risk assessment in the public process will be most severely tested. Can risks be presented in ways (e.g. comparatively) that can pave the way for their significance to be assessed properly, thus assuring that the evaluative stage is carried out in an appropriate way? Is the line between what the risks "are" and what they "mean" real or artificial?

The **evaluation** phase of the EIA process, made up of hearings, public review, and final decision-making, could profit from the use of risk assessment methods. As has already been mentioned frequently, there is a wide variety of notions of risk which serve a variety of purposes. This can be looked upon as a problem, or as an opportunity. One advantage of risk is that is that it creates a potentially common vocabulary for dealing with such diverse factors as scientific uncertainty, community concern, and economic projections. Carefully handled, risk can operate from a position of concern; which is, after all, the guiding principle of the EIA process.

The success of the **public participation** phase(s) of the EIA process may turn on the ability to analyse the perceptions (and the conceptions) of risk held by different individuals and sectors of a given community. Our understanding of the forces and mechanisms by which individuals, interest groups, proponents, and legislators interact to produce evaluations of risk acceptability is woefully inadequate. Failure to come to terms with the polarisations caused by concerns over risks and hazards confronts us with the prospect of continuing frustration and paralysis in many areas of environmentally-related decision-making.

The **monitoring** and **mitigation** phases of the EIA process can benefit from risk assessment simply because of the built-in iterative and hypothesis-testing aspects of risk methodology. Since prediction is central to this methodology, there is added incentive to monitor the results to see if, in fact, the "experiment" succeeded. Recently, EIA has come to see the phase after implementation as a key element in the overall success of coping with environmental impacts. Competent risk assessment may not only be worthwhile for each project, but the analytical framework developed for one EIA to assess the potential risks of new technologies, etc., need not be rediscovered at each succeeding EIA, but need only be refined and reshaped in light of new knowledge and the contingencies of each case.

This last matching simply underlines, once again, the view that the linearity of the EIA process from scoping through to monitoring and the RA process from risk identification through to final assessment should in fact be considered as circular or iterative. The RA process is particularly well suited to such a reconceptualisation, since the initial risk identification phase of that process is deeply embedded in the past history of coping with risks, and the slow convergence of probability distributions of greater and greater accuracy. This iterative view of environmental and risk assessment through time (diachronic) as well as in instantaneous snapshots (synchronic) sketches out a spiralling trajectory of greater sophistication with each iteration.

CONCLUSIONS

Risk assessment - and EIA - is in the business of handling the facts from a position of concern. The most developed parts of the risk field are in those areas in which the handling of facts is paramount; but even in these areas there is much research which needs to be done in order to achieve a level of sophistication at which the information gathered by risk analysis can be usefully applied by concerned citizens and public assessors mandated to serve the public good. In this area of evaluating the social significance of risk the available material is extensive, but scattered in several disciplines (e.g. anthropology and social psychology).

It is one of the implicit arguments of this volume that risk assessment and environmental impact assessment can be mutually supportive, and mutually illuminating. On the one hand, environmental impact assessment can be made more consistent by borrowing from risk; and on the other, risk assessment can be broadened by having to contend with the varieties of experience subsumed by an environmental impact assessment, and by having to satisfy the administrative procedures of a legally mandated planning instrument such as EIA.

REFERENCES

Beanlands, G.E. 1986. Risk in EIA: the perspective of the Canadian Federal Environmental Assessment and Review Office. Conceptual Trends and Implications for Risk Research, (Eds.) C.T. Miller, P.L. Kleindorfer and R.E. Munn. International Institute for Applied Systems Analysis, CP-86-26:53-59.

Beanlands, G.E. and P.N. Duinker. 1983. An ecological framework for environmental impact assessment in Canada. Institute for Resource and Environmental Studies, Dalhousie University.

Berger, Mr. Justice T.R. 1977. Northern frontier, northern homeland: the report of the Mackenzie Valley Pipeline Inquiry. Minister of Supply and Serives, Ottawa.

Bowonder, B. 1981, Environmental risk management in the Third World. Intern. J. Environ. Studies, 18:41-49.

Burton, I. 1982. The evolution of risk and risk assessment. Paper presented Statistical Soc. of Canada Meeting, Ottawa, July.

Burton, I. and A. Whyte. 1980. Environmental risk assessment. SCOPE 15. Wiley, New York.

Burton, I., C.D. Fowle and R.S. McCullough (Eds.). 1982. Living with risk: environmental risk management in Canada. Institute for Environmental Studies, University of Toronto, Toronto, EM-3.

Couch, W.J. (Ed.). 1983. Environmental assessment in Canada: 1982 summary of current practice. Canadian Council of Resource and Environment Ministers, Ottawa.

Dworking, R. 1978. Taking rights seriously. Harvard University Press, Cambridge, Mass.

Grima, A.P., P. Timmerman, C.D. Fowle and P.H. Byer. 1986. Risk management and EIA: research needs and opportunities. A background paper prepared for the Canadian Environmental Assessment Research Council. Minister of Supply and Services Canada, Ottawa.

Kates, R. 1978. Risk assessment of environmental hazard. SCOPE 8, Wiley, N.Y.

Krewski, D. and P.L. Birkwood. 1987. Risk assessment and risk management. Risk Abstracts, 4:53-61.

Miller, C.T. 1985. Report on risk management workshop. Background paper for the Inquiry on Federal Water Policy.

Rogers, J.T. and D.V. Bates (Eds.). 1983. Proc. Symp. Assessment and Perception of Risk to Human Health in Canada, Royal Society of Canada, Ottawa.

Ronge, V. 1982. Risk and the waning of compromise in politics. In: Risk Analysis Controversy: An Institutional Perspective, (Eds.) H.C. Kunreuther and E.V. Ley. Springer-Verlag, Berlin.

Rowe, W.D. 1977. An anatomy of risk. Wiley, New York.

Starr, C. 1969. Social benefit versus technological risk. Science, 165:1232-1238.

Timmerman, P. 1984. Ethics and hazardous waste facility siting. Institute for Environmental Studies and Department of Civil Engineering, WM-84-11, University of Toronto, Toronto

Whitney, J.B.R. and V.W. Maclaren. 1985. A framework for the assessment of EIA methodologies. In: Environmental Impact Assessment: the Canadian Experience, J.B.R. Whitney and V.W. Maclaren. Inst. for Environmental Studies, University of Toronto, Toronto, EM-5:1-32.

CHAPTER 2
THE APPLICATION OF RISK ASSESSMENT IN CANADIAN ENVIRONMENTAL IMPACT ASSESSMENT

P.J. Paradine
Environment Sector, Canadian International Development Agency

INTRODUCTION

The past decade has seen increasing use of risk assessment in environmental impact assessment (EIA) procedures in Canada, particularly in the management of a number of environmental hazards to people and the environment.This chapter describes a number of cases to illustrate the use of risk assessment in EIA within Canada in the late seventies and early eighties.

EIA procedures are used to identify, predict and resolve environmental concerns that may be associated with a proposed project. The process requires prediction of likely consequences, together with recommendations on mitigation or compensation. Procedures in Canada make extensive use of public involvement as well as technical review of studies prepared by experts. (Couch (1983) describes EIA procedures at the federal level as well as in all ten provinces.) Hazards to humans are normally considered as an integral part of the environmental impact assessment and the socio-economic impacts of projects are increasingly being taken into account.

For our purposes, risk assessment (RA) encompasses both risk analysis (i.e. the estimation of the probability of a given event) and risk evaluation, i.e. assessing the significance of the consequences of such an event in terms of its potential social, economic and environmental impacts. Canadian EIA procedures appear to readily incorporate these risk assessment concepts. Both RA and EIA share the challenge of identifying the hazardous event and its consequences, from the perspective of scientists and lay people. Other common challenges include the separation of natural fluctuations from anthropogenic perturbations; consideration of natural and man-made risks; indirect and latent effects; and all the complexities associated with arriving at acceptable trade offs (e.g. by carrying out a benefit-cost analysis or a multi-attribute utility analysis).

Indeed, the EIA process itself may be considered a form of risk assessment, with hearings providing a public perspective so necessary to the decision-maker. The key to risk assessment and risk analysis is the estimation of the probability of an event occurring. In EIA there has long been a lack of quantification, the need for which has been reaffirmed in the Canadian context by Beanlands and Duinker (1983). The increasing application of risk assessment in EIA has, therefore, required a somewhat more quantitative approach in addressing risks to human health and environmental quality. The use of risk assessment in EIA is becoming more common and more explicit. This paper examines and compares some of these applications and analyses their utility in the decision-making process.

HYDROCARBON PROJECTS

In Canada risk assessment principles have been most often used in environmental impact assessments for hydrocarbon exploration, production and transportation projects. Risk assessment was explicitly used in five proposed offshore exploration projects in Canada: South Davis Strait; Lancaster Sound; offshore Labrador; Sable Island and off the West Coast. Other examples of the use of risk assessment include the Venture and Beaufort Sea production projects.

Risk assessment (RA) in a federal EIA was first required in the guidelines for preparing an environmental impact statement (EIS) for eastern Arctic exploratory drilling (FEARO 1978) and such a requirement for RA appeared in all subsequent federal offshore projects. In this case the guidelines required a risk analysis of a blow-out, including estimates of the probabilities of spills of various sizes; an oil dispersion model; methods for estimating the trajectory and dispersion of any release of sour gas; and a discussion of the hazards of pack ice, icebergs and drilling systems.

The first project reviewed in accordance with these guidelines was a proposal by Imperial Oil et al. (1978), to drill from a drill-ship in the South Davis Strait, southeast of Baffin Island. The major threat identified in the EIS was an uncontrolled blowout with an estimated probability of one in 1,000,000; the EIS by Imperial Oil et al. (1978) also pointed out that 90% of blowouts involve only gas.

The threat of a blowout emerged as the major concern of intervenors during public meetings in Baffin Island communities. The Panel was presented with probability estimates that ranged from three in 10 for water blowouts (Beaufort Sea), none in 125 (east coast offshore wells) and down to one in 3 million (world-wide experience for oil blowouts).

The Panel's report concluded that the probability of an oil blowout was low and the likely impacts from even a "worst-case" would not be serious (EAP 1978). The overall conclusion was that the environmental risk of the project was acceptable provided that certain conditions were followed. One well was subsequently drilled without incident.

The guidelines for the eastern Arctic were applied to the Norlands Petroleum's (1978) proposal to drill an exploratory well in Lancaster Sound. The proponent's EIS included a worst-case scenario of 6,000 barrels of oil per day flowing for a full year. Some intervenors felt that the project would almost certainly result in a major oil blowout; others considered the possibility of a blowout so remote that it did not merit discussion. The Panel report noted that generally speaking the probability of major oil blowouts for offshore drilling is minimal; however, the study area was infested with ice and the drilling season is very short (EAP 1979).

The Panel's conclusion was that a realistic appreciation of the severity of physical hazards was necessary. Although the probability was relatively low, a blowout presented the greatest threat to the environment and major damage could result. The Panel recommended against drilling approval until the proponent demonstrated a capability to deal safely and effectively with the physical hazards in Lancaster Sound, and an operational preparedness to mitigate the effects of a major oil blowout.

The four Initial Environmental Evaluations (IEE) produced for offshore hydrocarbon exploration also addressed the issue of blowouts during exploratory drilling. The first, Mobil (1980), estimated a world-wide average probability of two blowouts per 1000 wells and discussed the Ixtoc, Ekofisk and Santa Barbara incidents.

The next IEE to be completed was for offshore Labrador. Petro Canada (1982) concluded that the probability of an oil blowout was very low and that there was little to be gained by estimating exact probabilities, other than better design of offshore drilling systems. Eventually, 24 wells were drilled off Labrador and no blowouts occurred.

An IEE for exploratory drilling offshore of the Queen Charlotte Islands (Petro Canada 1983) maintained that too few wells had been drilled to calculate the risk of a blowout precisely and to assess its consequences. Human error was identified as a primary risk factor.

In its IEE for exploratory drilling on the northern coast of British Columbia, Chevron (1983) also states that the probability of a well blowout is extremely small. This conclusion is based on reports of an average of one blowout for every 300 wells drilled offshore, one "major" blowout in 3000 offshore wells and no blowouts resulting in an oil spill during exploration drilling. These estimates, however, depend upon the designation of the Ixtoc well as non-exploratory.

The Venture Gas Offshore project off Nova Scotia was the first formal EARP review of a **production** proposal to be completed, and risk assessment figured prominently in the report by the Federal-Provincial Environmental Assessment Panel (1983). The Panel noted that there was a significant possibility of a well blowout during development and production of the Venture field and that this could result in major fire hazards as well as environmental impacts. The proponent's EIS (Mobil 1983) stated that 97.5% of all wells in operation in recent years were incident-free. Mobil estimated the chance of a blowout to be 15% (with a fire 3.3%). Two months after release of the Panel report, the first blowout on the east coast occurred at a nearby exploration well.

Risk assessment was a major issue during the federal review of the Beaufort Sea Project. This project involved an area-wide EIS (Dome et al. 1982) for hydrocarbon production from Beaufort Sea reserves. Transportation alternatives included a pipeline down the Mackenzie Valley or ice-breaking oil tankers operating through the Northwest Passage.

The assessment of oil spill risks for the Beaufort Sea proposal included an analysis of historical data from various oil-producing regions of the world, of different types and ages, modified to reflect Arctic conditions and up-to-date techniques. There was considerable debate over methodological issues but the Panel accepted the view that it was possible to arrive at reasonable bounds for spill frequency and size provided that judgement was used in the interpretation of the analysis. Because of the conceptual nature of the review, various production rate scenarios and transportation alternatives were considered. It was noted that probabilities would increase at higher production rates, but average spill sizes would not likely increase. Maximum "credible" spill sizes over the first 20 years of Beaufort Sea production were compared with the largest oil spills recorded in the world (see Table 1). Risks were estimated for

Table 1. Maximum credible spill sizes estimated for various components of the proponents' proposed production and transportation systems

System component	(1) Maximum credible spills at about 15,900 m^3/day production level	(2) Maximum credible spills at 110,000 m^3/day production level	(3) Maximum recorded oil spills sizes in world[1] data for comparison
	m^3	m^3	m^3
1. Development drilling			
non-blowout	-	-	50
blowout	330,000	330,000	490,000
2. Production operations			
non-blowout	-	-	240
blowout	-	-	24,000
3. Pipelines			
subsea	95	750	25,000
overland	1,100	5,600	9,500
4. Tankers			
collision, ramming, grounding	41,000	41,000	-
structural failure explosion, fire	220,000	220,000	240,000
5. Offshore storage for tankers	43,000	43,000	120,000

[1] World statistics do not differentiate very well between production and development blowouts. Although 330,00 m^3 was given as the maximum credible spill associated with development and production combined, the spill associated with production operations should be smaller.
[2] Cubic metres may be converted to barrels by multiplying by 6.28.
Source: Lemberg (1983).

various components of the overall project, including development drilling and production operations (blowouts and non-blowouts), pipelines (subsea, overland and storage) and tankers (harbour, restricted waters, open sea and storage).

The proponents maintained that tankers would have risk characteristics comparable to those of production and development drilling; spill probabilities and maximum volumes would be lower. These estimates assume the use of specially-constructed tankers for use in the Arctic; in fact the system proposed was considered to be many times safer than that of a conventional tanker in the south. Risk analysis figured prominently in the Panel's report (EAP 1984). The Panel concluded that the risk associated with the project should be acceptable to residents of the region.

HYDROCARBON TRANSPORTATION PROJECTS

Several hydrocarbon transportation projects have been subject to comprehensive review: three liquified natural gas (LNG) proposals, and the transportation components of the Beaufort Sea and Venture Gas production proposals. LNG proposals in Quebec (Gross Cacouna) and Nove Scotia (Melford Point) were reviewed by federal-provincial panels. The panel reports, EISs and transcripts of public hearings make extensive reference to risk assessment. Proposals for a LNG terminal in British Columbia were subject to review by the British Columbia Utilities Commission (BCUC). The Port Simpson (NWT) Dome proposal that was submitted to the N.E.B., also includes an extensive section on risk assessment.

In March 1981, a federal-provincial Panel approved a liquified natural gas terminal at Gros Cacouna on the Gulf of St. Lawrence (BAPE 1981). The Panel concluded that the risks were acceptable considering the criteria prevailing in Canadian society. However, the Panel recommended further studies on seismicity and recommmended certain safety measures. The liquified natural gas was to be shipped in ice-breaking tankers as part of the Arctic Pilot Project.

The Panel report (EAP 1981) on the Melford Point LNG in Nova Scotia addressed the risks associated with a shipping route up to 60°N as well as with the terminal and its approaches in the Strait of Canso. The Panel concluded that the LNG ships would pose no significant increase in risk to fishing operations or small vessels, and that the project could be constructed and operated within an acceptable level of risk for persons living along the proposed shipping route.

During the public meetings, the proponent's risk analysis was the subject of discussion by review agencies and a technical expert hired by the Panel. In the Panel's report, it was suggested that the proponent's risk analysis could be improved and should be re-examined to further increase its reliability and thus enhance confidence in the overall safety of the project.

Dome's application to the National Energy Board for a proposed LNG facility at Port Simpson includes its risk analysis (Dome 1983). The public risk of a fatality is assessed at between $1x10^{-8}$ and $1x10^{-9}$ per year as compared with an occupational risk of fatality of $1.3x10^{-4}$ per year. These risks were compared with the risk of being struck by lightning ($1x10^{-6}$) and the average industrial risk of an employee fatality for Canada ($1x10^{-4}$). The risk of an LNG carrier colliding with a vessel was assessed at $1.6x10^{-6}$ and compared with aircraft collisions ($1x10^{-5}$). The only credible worst spill scenario envisioned was a carrier in collision with a ferry or cruise ship, since fishing vessels were not seen as being able to breach the LNG carrier. Several approach routes were analysed and the report identified those that reduced risks. Preferable passing zones and vessel traffic patterns were also suggested.

A comparative risk assessment on Pacific coast oil ports was prepared by Fisheries and Environment Canada (1978). This was originally prepared for presentation to a federal enquiry which started in 1977. However, the proposal and the inquiry did not proceed and the analysis of Millen and Sherwood (1979) was eventually used in an NEB hearing on the Transmountain Project. The latter would have taken oil shipped from

Valdez, Alaska to Port Angeles in Washington State (just south of Victoria), and then through Canada by pipeline to U.S. northern tier states. Since the oil would have been shipped past the British Columbia coastline, there was considerable concern over the possibility of a tanker spill even though the proposed routings could potentially have reduced risk levels. The Canadian Coast Guard, through its Termpol Committee, also prepared a report on the navigational risks associated with Canadian ports, including Kitimat versus U.S. ports (Transport Canada 1979). No definite project emerged and there was no formal EIA review, but the preparatory documentation developed included an extensive application of risk assessment to EIA.

SOUR GAS PROJECTS

Sour gas facilities in Alberta have also been subject to risk analysis in EIA (Angle 1982). The directive for siting (ERCB 1981) requires minimum distances separating new sour gas facilities from residential and other developments. There is still provision in the directive for case-by-case analysis of potential release volumes over 6000 m^3, and large potential impacts have been cited by the Board as grounds for not approving certain facilities. Proposed development in the Kananaskis area of Alberta was initially rejected because of possible effects on recreational users, but the decision was overturned on appeal following further work on gas dispersion models by Alberta Environment (ERCB 1983).

The sour gas blowout in Lodgepole, Alberta, was the subject of an inquiry by the ERCB (1984). Phase II of the inquiry addressed the improvements that might be made in drilling operations to reduce the risk of a similar blowout. The ERCB report provided an analysis of blowouts (one in 172 for sour gas, one in 2900 for all wells drilled in Alberta) and includes numerous recommendations to reduce sour gas blowouts with special emphasis on reducing human error (Rogers 1988).

NUCLEAR POWER PROJECTS

The Atomic Energy Control Board requires that regulatory agencies undertake a risk assessment for nuclear power projects as part of the licensing procedure. Perhaps because Lepreau is the only Canadian nuclear generating station whose EIS was subject to public review, EIA experience in this area is not as extensive as that of the United States. The original station (Lepreau I) was the subject of a federal-provincial report (EAP 1975) which discussed radioactivity hazards, including routine emissions and malfunction and expressed confidence in the AECB's regulatory approach to address these issues. The proposed second reactor (Lepreau II) underwent a review by another Panel that issued EIS guidelines for a risk assessment (Federal-Provincial Environmental Assessment Panel 1984).

The Panel has also requested estimates of the probability of upset conditions, (e.g. earthquakes) that would result in the release of radionuclides.The Lepreau II EIS (Maritimes Nuclear 1984) included discussions on safety design and analysis, enviromental radiation, decommissioning, transportation and long-term disposal of used fuel, emergency planning and the listing of postulated accidents all of which form the basis of the safety analysis requirements.

Discussion in the EIS on the releases of radioactivity to the environment deals with the concepts of derived emission limits (DEL) at levels which provide an adequated measure of protection for people. Radiation exposures to members of the public from normal operation of nuclear power plants are considered so low as to be obscured by normal fluctuations in natural background. Therefore, radioactive limits are established at source and DELs are calculated using various critical pathways. Some examples are pasture - cow - milk - infant - thyroid (for gaseous emissions) and sea food-fisherman families (for liquid effluents), clam diggers and dulse-gathers (for external exposure). The EIS maintains that radiation exposure from the two reactors at Lepreau would correspond to 0.6% of natural background; by comparison, exposure from the fallout from the 1980 Chinese nuclear weapons tests was several orders of magnitude greater.

The discussion on potential health risks from exposure to radiation assumed a linear dose-response relationship, i.e. an exposure of one sievert (SV) would yield a cancer risk of one in 100 (1% per SV). Based on estimated Lepreau figures, this is considered to result in an incremental cancer risk of 1.2×10^{-5} per year or 10^{-3} per lifetime for the most exposed individuals (i.e. those on the critical pathways). This was compared with the current 20-25% risk of dying of cancer, and the risk of dying from smoking one cigarette/year (1×10^{-5}).

FORESTRY PROJECTS

In general, EIA has been more comprehensively applied to specific projects rather than programs such as the use of chemicals in forestry management, a topic which has been the subject of considerable public concern. Various institutional approaches have been used by governments to assess the health concerns raised by the application of substances such as 2,4,5-T, or spraying to control spruce budworm infestation. In New Brunswick, a task force was formed to assist the government in deciding on spruce-budworm spraying. In Newfoundland, a Royal Commission was appointed to conduct a similar task.A Nova Scotia Supreme Court judge ruled in 1983, that spraying of 2,4,5-T could be permitted on certain forestry lands, and the results of a risk assessment figured prominently in the reasons for the court's decision. However, Quebec is the only province where an EIA is required for programs such as spraying of forest land for spruce-budworm control. In 1982, Bureau d'Audiences Publique sur l'Environnement (BAPE) held public hearings and reported to Cabinet on a proposed spraying program. An EIS was prepared to consider the environmental health and socio-economic effects of a further long-term spray program and further public hearings were held. Risk assessment was used extensively in the EIS.

The EIS (Marsan 1984) developed a suggested strategy for control of spruce budworm through forestry management and spraying during the period 1985 to 1989. The socio-economic implications of various options were studied as was the risk due to spraying. The recommended approach was a blend of salvage, reforestation and spraying with aminocarb, fenitrothion and a progressively increasing use of <u>Bacillus thuriengensis</u> (B.t.).

The toxicology of the three chemicals was researched to determine a no-effect dose, and the worst-case exposure of people and organisms

calculated using a mathematical model calibrated with field measurements. It was estimated that the inhalation and ingestion exposures would be 100 times less than the no-effect dosage. Adding exposure received through skin contact, the worst case situation was estimated to be between three and 10 times less than no-effect dosages, using conservative assumptions (e.g. extended presence in the area being sprayed). Reference was made to experiences elsewhere, including epidemiological studies in New Brunswick. The EIS considered the impacts on the mortality of pollinators and birds, and calculated the risks of toxicological effects for a variety of other wildlife.

B.t. was considered to be a completely inoffensive spraying product. However, its effectiveness is still seen as experimental, so that its use as the sole spraying product in the initial stages of the program was not recommended, but a complete changeover to B.t. by 1988 was suggested. The EIS recommended phasing out all spraying with synthetic organic chemicals eventually because of the uncertainty over long-term effects.

OTHER APPLICATIONS

Incidents such as the Mississauga train derailment have demonstrated risks involved in shipment of hazardous chemicals by rail. Where the infrastructure is already in place, EIA is not likely to be applicable, since EIA procedures apply to only new projects. Risk assessment, however, is used by companies and regulatory agencies concerned with safety on an ongoing basis. Recent efforts to expand railway capacity through the Rocky Mountains has resulted in some EIA reviews involving transportation risk assessment. In the case of transportation down the Fraser River Canyon, a risk assessment of existing traffic has been carried out by the Environmental Protection Service of the Department of the Environment (Millen 1980). This analysed the number of accidents on CN tracks and pin-pointed key problem areas for derailment and the amount of various hazardous substances potentially involved. Chlorine was identified as a significant problem. Plans to twin-track the CN line resulted in the appointment of an Environmental Assessment Panel. Risk assessment was identified as an issue, and the Panel in its interim report (EAP 1983) requested information on what analysis had been carried out by CN, the potential high risk areas, the means of reducing the risks and what emergency response programs currently exist. Further discussion on these topics has took place during public meetings.

Flood frequency analysis was the basis for a discussion of risks during public hearings on the Mille Iles project. The project was designed for the protection of an area around the Island of Montreal. BAPE (1981) reported that the information presented to it was insufficient to make recommendations.While some concerns have been expressed during public hearings on failure of dams in B.C. (Revelstoke) and Labrador (Lower Churchill), risk assessment has not been extensively applied in EIA reviews on such projects.

CONCLUSIONS

The most frequent application of risk assessment in EIA involves high consequence, low probability occurrences (LOPHIC) which are the kinds of risks that the general public is most concerned about. Public perception and concern are addressed in the EIA through public meetings. The review process provides an opportunity for useful exchanges between the public and technical agencies, during which the assumptions on which predictions are made can be questioned. The conclusions, with the necessary reservations and qualifications, are then presented to decisonmakers, who have to evaluate the various risk estimates. Through EIA the risk associated with development projects has been subjected to vigorous debate, albeit less frequently than consequences that are more routine in EIA.

In some cases, review bodies have examined a range of values for the estimated probability of an event, before concluding that the acceptability of the proposal is not prejudiced by the differences in expert opinion. In other cases, such as offshore drilling, the refinement of information through a number of reviews of similar projects has resulted in substantially increased probability estimates. The acceptability or non-acceptability of such projects does not seem to have been altered by the changing probabilities.

It appears, that while the probability of occurrences is of value in the decision-making process, the nature and magnitude of the consequence of events are more critical. That is one explanation for the concern over worst case scenarios during public reviews. A more useful effect of risk assessment in EA has been in identifying the causes of risk and the areas in which mitigation is feasible. Notwithstanding the limitations of data bases and the differences in methodologies, the debates on the characterization and estimation of risk has enhanced the EIA process, and provide a testing-ground for working hypotheses. Although the quality of risk assessment within EIA has varied widely, the refinements that have occurred over a series of reviews, in particular for hydrocarbon developments, suggest that increased use of risk assessment improves the information input to the decision making process. For these reasons increased use of risk assessment in EIA is to be encouraged.However, further research is required on how risk assessment might be better integrated in Canadian EIA processes.

A particular difficulty in the use of risk assessment in EIA has been the variety of analytical methodologies employed, and the lack of agreement in the application of the various approaches. The studies presented here have demonstrated a range of approaches. Examples can be found of risk-benefit (i.e. spruce budworm spraying, sour gas), risk-comparison (nuclear power, LNG) and relative risk (West coast LNG proposals). Further codification of methodologies is needed (see also Grima et al. 1986). It could be argued that the use of risk assessment, particularly the sophisticated use of mathematical models, statistical methods and data bases, has tended to restrict the field to specialists. In some cases a bewildering range of numbers have been produced; on occasion a credibility problem may have been caused, as statistics have produced conflicting conclusions. Where no-risk contentions are advanced, deeper philosophical conflicts between the parties are indicated. The experience of EIA may offer some solace in that no "magic" methodology

exists and indeed, debate is encouraged. Since the estimated probability of an event appears to be less critical to public concern and decision making, this lack of agreement on estimation methodology does not seem to present major practical problems. On the other hand the type and magnitude of consequences of predicted events have played a crucial role in EIA decisions. Equally important has been the focus on risk reduction. On a number of occasions the absolute numbers have been discounted in favour of identifying ways in which risk could be lowered by use of appropriate, or better technologies. In this respect risk assessment seems to be increasingly instrumental in achieving more environmentally acceptable development.

REFERENCES

Angle, R.P. 1982. Sour gas facilities a case study of public risk in Alberta. In: Living with Risk: Environmental Risk Management in Canada, (Eds.) I. Burton, C.D. Fowle and R.S. McCullough, Institute for Environmental Studies, University of Toronto, EM-3:105-115.

Beanlands, G.E. and P.N. Duinker. 1983. An ecological framework for environmental impact assessment in Canada. Institute of Resource and Environmental Studies, Dalhousie University, Halifax, N.S. and FEARO, Hull, P.Q. 132 pp.

Bureau d'Audiences sur l'Environnement (BAPE). 1981. Projet de terminal methanier a Gros Concouna. Rapport d'Enquete et d'Audience Publique, Quebec.

Bureau d'Audiences sur l'Environnement (BAPE). 1982. Projet d'ouvrage de controle sur la Riviere des Milles Isles. Quebec.

Chevron. 1983. Initial environmental evaluation of renewed petroleum exploration in Hecate Strait and Queen Charlotte Sound. FEARO, Ottawa.

Couch, W.J. 1983. Summary of current practice, EIA in Canada. Canadian Council of Resource and Environment Ministers, Ottawa.

Dome Petroleum, Esso Resources, Gulf Resources. 1982. Beaufort Sea-Mackenzie Delta environmental impact statement. FEARO, Ottawa.

Dome Petroleum. 1983. Western LNG project, application to the National Energy Board. FEARO, Ottawa.

Energy Resources Conservation Board (ERCB). 1981. Directive 81-3. Edmonton.

Energy Resources Conservation Board (ERCB). 1983. Shell's proposed Moose and Whiskey pipeline and related facilities, Kananaskis area. Edmonton.

Energy Resources Conservation Board (ERCB). 1984. Lodgepole blow-out inquiry, Phase II. Edmonton.

Environmental Assessment Panel (EAP). 1975. Report on Point Lepreau nuclear generating station. FEARO, Ottawa.

Environmental Assessment Panel (EAP). 1978. Eastern Arctic offshore drilling - South Davis Strait project report. FEARO, Ottawa.

Environmental Assessment Panel (EAP). 1979. Lancaster Sound drilling report. FEARO, Ottawa.

Environmental Assessment Panel (EAP). 1981. Report on liquified natural gas receiving terminal Arctic pilot project, Melford Point, Nova Scotia. FEARO, Ottawa.

Environmental Assessment Panel (EAP). 1983. Interim Report on CN rail twin tracking program, B.C. FEARO, Ottawa.

Environmental Assessment Panel (EAP). 1984. Report on Beaufort Sea hydrocarbon production and transportation. FEARO, Ottawa.

Federal Environmental Assessment Review Office (FEARO). 1978. Guidelines for preparation of an environmental impact statement for Eastern Arctic exploration drilling. FEARO, Ottawa.

Federal-Provincial Environmental Assessment Panel. 1983. Venture development project report. FEARO, Ottawa.

Federal-Provincial Environmental Assessment Panel. 1984. Guidelines for the preparation of an environmental impact statement, Lepreau II. FEARO, Ottawa.

Fisheries and Environment Canada. 1978. Potential Pacific coast oil ports, a comparative environmental risk analysis. Ottawa.

Grima, A.P., P. Timmerman, C.D. Fowle and P. Byer. 1986. Risk management and EIA: research needs and opportunities. Canadian Environmental Assessment Research Council, Ottawa.

Imperial Oil, Aquitaine, Canada City Services. 1978. Environmental impact statement for exploratory drilling in South Davis Strait. FEARO, Ottawa.

Lemberg, R. 1983. Simplified summary of oil spill risk assessment. FEARO, Ottawa.

Maritime Nuclear. 1984. Lepreau II environmental impact statement. FEARO, Ottawa.

Marsan et Associees. 1984. Evaluation environnementale et economique, programme de Pulverisations Contre la Tondeuse des Bourgeons d'Epinette.

Millen, J. 1980. Environmental risk assessment of the transport of dangerous goods on the CNR mainline in B.C. FEARO, Ottawa.

Millen, J. and W. Sherwood. 1979. Comparative environmental assessment of west coast oil transport alternatives. FEARO, Ottawa.

Mobil Oil Canada Ltd. 1980. Initial environmental evaluation for determination drilling, Sable Island area.

Mobil Oil Canada. 1983. Venture development project environmental impact statement.

Norlands Petroleum. 1978. Environmental impact statement for exploratory drilling in the Lancaster Sound region.

Petro Canada. 1982. Offshore Labrador initial environmental assessment.

Petro Canada. 1983. Offshore Queen Charlotte Islands, initial environmental evaluation.

Rogers, R.E. 1988. The Lodgepole sour gas well blowout. In: Information Needs for Risk Management, (Eds.) C.D. Fowle, A.P. Grima and R.E. Munn. Institute for Environmental Studies, University of Toronto, EM-8:205-221.

Transport Canada. 1979. Assessment of ship accident and oil spill risks associated with potential West Coast oil ports. FEARO, Ottawa.

CHAPTER 3
THE WORST CASE RISK ASSESSMENT REQUIREMENTS OF THE U.S. NEPA PROCESS

W.D. Dickerson,
U.S. Environmental Protection Agency, Washington, D.C.

INTRODUCTION

BACKGROUND

Notwithstanding the fact that the U.S. National Environmental Policy Act (NEPA) has been around for almost 20 years, the application of formal risk assessment (RA) techniques to the analysis of impacts on the natural environment has just begun to emerge. This recent emergence has been influenced by both the development of risk assessment methods for human health impact analyses, as used by U.S. regulatory agencies, and the recent controversy over the worst case provision of the NEPA implementing regulations. This provision has, unfortunately, been interpreted by many as the key provision in the NEPA implementing regulations that require risk assessments to be done in certain Environmental Impact Statements (EISs). In fact, the general requirements of the regulations provide for the use of risk assessment techniques and the worst case provision simply adds requirements if information is missing or incomplete.

Recent court cases involving application of the worst case provision prompted significant public debate on what this section meant and what it should do. The outcome was a revision of the worst case regulation to clarify its intent. Since this provision has been a focal point for risk assessment applications in the NEPA process, I believe that some of the ideas raised in this debate may be useful as Canada and others implement RA in their environmental assessment (EA) processes. In this paper, I present a brief summary of the evolution of the risk assessment process in the U.S. as applied to human health impacts from toxic chemicals and with that as background, describe how this human health based approach to risk assessment fits within the provisions of the U.S.NEPA process.

My belief is that the development of RA methods for these different applications has been synergistic but that the state of the art in applying risk assessment techniques to impacts on the natural environment still has a long way to go. The good news is that the human health based RA **principles** that have been developed are applicable to the NEPA process and should help clarify analytical development in this area. The bad news is that some of the specific analytical requirements (such as developing dose-response data for ecosystems) are very difficult, if not impractical or impossible, to obtain and will, therefore, require a rethink of the quantitative versus qualitative "mix" of RA assessment techniques applicable to the analysis of natural systems.

RA DEVELOPMENT IN THE U.S.

Risk assessment work related to human health began to get serious in the U.S.in the early 1970s when the U.S. Food and Drug Administration (FDA) had to deal with regulating the chemical food additive DES (Diethylstilbesterol) under the so-called "Delaney" clause in the Food,

Drug and Cosmetics Act as amended in 1968. This clause established a prohibition against the addition of carcinogenic chemicals into food for livestock if there would be detectable residues of the carcinogen in the food-producing animals at the time of slaughter. FDA chose to deal with this problem in terms of a quantitative risk assessment approach rather than the traditional "zero detection within the sensitivity of the analytical method" approach (Flamm 1985).

After FDA's initial work, the Occupational Safety and Health Administration, Consumer Product Safety Commission and the Environmental Protection Agency also began using RA techniques to support their regulatory activities.

In 1981, Congress directed the National Academy of Sciences (NAS) to conduct a study of the institutional process for risk assessment. This effort resulted in their well known report, "Risk Assessment in the Federal Government: Managing the Process" (U.S. National Research Council 1983).

This report was clearly a major milestone in the risk assessment development process in that it provided a common set of terms that the various agencies could utilize in their RA development work. It adopted a two part approach in describing the RA process. First it separated risk assessment from risk management. The report defined risk assessment as the characterization of the potential adverse health effects of human exposure to environmental hazards, where "characterization" includes consideration of the uncertainties inherent in the process of measuring and evaluating risk. NAS also emphasized that risk assessment must allow for the use of both quantification techniques and qualitative expressions of risk. Second, and probably more important, the NAS defined risk assessment in terms of a four step process: hazard identification; dose-response assessment; exposure assessment; and risk characterization.

After publication of the NAS report, then EPA Administrator, William Ruckelshaus started a major effort to bring some uniformity and credibility to EPA's risk assessment/risk management processes (U.S. EPA 1984). His concern was that the presentation of the technical analyses and the folding in of other considerations was not consistent throughout the Agency. Administrator Ruckelshaus felt strongly that since EPA had to make decisions on the basis of analyses provided by technical people, the information had to be presented in a form that could bridge the gap between the technical and policy levels. He also recognized that, in the end, Federal decision-makers must, for the most part, make a "judgemental" decision that cannot be totally explained or justified to the public that must live with that decision. For that reason, he believed that the "trust" of the public in the objectivity of the decision maker is critical. To gain that trust, he said, requires open and full disclosure of the facts, methods, and uncertainty in reaching these decisions (U.S. EPA 1984). The evolving RA and risk management methodologies were to be key factors in accomplishing these goals.

EPA subsequently adopted the NAS framework and began to develop and publish risk assessment guidelines related to cancer risk (U.S. EPA 1986). In addition, the agency established a risk assessment forum to discuss risk assessment and science policy issues of concern to all offices in the Agency (Ehrlic 1985).

On a national level, the Office of Science and Technology Policy was established as an Executive Office of the President and began to provide a focal point for RA development, which was now going on in over a dozen federal health related agencies. In 1985, that office published a two-part national report which:

1. presented a set of general principles for use by regulatory agencies as they review their own specific guidelines for performing cancer risk assessments; and

2. described the current state-of-the-science concerning carcinogenesis and cancer risk assessments (US Office of Science and Technology Policy 1985). This report also utilized the four risk assessment steps laid out in the NAS report as a framework for their discussion.

In the U.S. then, the NAS principles for describing the RA process associated with human health impacts have become firmly entrenched.

U.S. REGULATORY STRUCTURE FOR RA IN EA

The U.S. NEPA process is essentially procedural in nature with enforcement of these procedures left up to citizens to pursue through the courts. This has lead to a heavy emphasis on the "legal" rather than the technical defensibility of analyses contained in many U.S. EISs and, in fact, drove the debate on the worst case revisions. This section describes the overall NEPA regulatory structure and shows how RA techniques are included in the overall analytical requirements of the U.S. implementing regulations.

GENERAL STRUCTURE REGULATING EIS PREPARATION

The U.S. procedural structure for implementing NEPA consists of the following elements:

NEPA: Establishes the legal requirement for federal agencies to incorporate environmental considerations in their decision-making, requires the "action forcing" preparation of Environmental Impact Statements, and creates the Council on Environmental Quality (CEQ) to, among other things, oversee the implementation of the NEPA requirements. NEPA has not been substantially amended since it became law January 1, 1970.

CEQ Regulations: The CEQ first published guidance to federal agencies for implementing the NEPA requirements in April 1973. Since this guidance was not a binding requirement on the agencies, federal compliance was not very successful. As a result, on May 4, 1977, President Carter signed Executive Order (EO) 11991 (Amending EO 11514) which required CEQ to publish binding regulations governing the federal implementation of NEPA and reaffirmed that federal agencies were to develop associated detailed procedural guidance specific to their activities. CEQ published their "Regulations for Implementing the Procedural Provisions of the National Environmental Policy Act November 29, 1978" (CEQ 1978). CEQ has also issued non-binding guidance since then to clarify the regulations

(CEQ 1981; 1983a). One of these, their "40 Most Asked Questions" guidance (CEQ 1978), dealt with the worst case provisions, among other things.

Agency Specific NEPA Guidance: Each federal agency has prepared, in consultation with CEQ, other federal agencies, and the interested public, specific detailed procedural guidance for carrying out their NEPA requirements. The level of detail varies considerably from agency to agency but the guidance generally describes the process they follow in determining if an EIS is needed and the process for coordinating and preparing EISs.

Agency Specific Supplemental Guidance: Many agencies have prepared additional guidance which gives specific information on how to carry out procedural requirements and technical analysis. The larger construction oriented agencies are also involved with outside groups which foster technical oversight and development. For example, the U.S. Federal Highway Administration works closely with the Transportation Research Board and US Civil Engineering Societies and the U.S. Minerals Management Service has formed both policy and science advisory boards to advise them on matters related to offshore oil and gas development.

SPECIFIC REGULATION REQUIREMENTS RELATED TO RA

The "definitive" requirement for applying RA under NEPA is contained in the CEQ regulations as part of the requirement for agencies to conduct environmental assessments. NEPA sets the framework for a full and rigorous analysis by directing that all agencies of the federal government must utilize a systematic, interdisciplinary approach to planning and decision-making, and that they identify and develop procedures which will ensure that presently unquantified environmental amenities and values are given appropriate consideration in decision-making.

The CEQ regulations further expand this framework for applying decision-oriented techniques (such as RA) in Sections 1501.2(b) and (c) by requiring that federal agencies: "Identify environmental effects and values in adquate detail so they can be compared to economic and technical analyses" and that they "study, develop, and decribe appropriate alternatives to recommended courses of action in any proposal which involves unresolved conflicts concerning alternative uses of available resources". Finally, in describing the purpose of the EIS, the regulations state in Section 1502.1 "...it (the EIS) shall provide full and fair discussion of significant environmental impacts and shall inform decision makers and the public of the reasonable alternatives which would avoid or minimize adverse impacts or enhance the quality of the human environment".

It seems clear that within this context, a complete NEPA analysis must include appropriate "risk analyses" as well as other assessment methodologies.Unfortunately, the risk analysis concept, within the NEPA process became identified with the worst case requirement of Section 1502.22. This Section, in reality, implements Section 102.2(b) of NEPA which deals with the situation of incomplete or unavailable information. Section 1502.22 of the 1978 CEQ Regulations stated, in Part:

> "(b) If (1) The information relevant to adverse impacts is essential to a reasoned choice among alternatives and is not known and the overall costs of obtaining it are exorbitant; or

(2) The information relevant to adverse impacts is important to the decision and the means to obtain it are not known (e.g. the means for obtaining it are beyond the state of the art), the agency shall weigh the need for the action against the risk and severity of possible adverse impacts were the action to proceed in the face of uncertainty. If the agency proceeds, it shall include a worst case analysis and an indication of the probability of its occurrence."

During June and July of 1980 the CEQ held a series of meetings with federal, state and local officials involved in the NEPA process. At these meetings the CEQ discussed, among other things, common questions about NEPA and the practical application of the NEPA regulations. As a follow-up to these meetings, CEQ compiled and published forty of the most important and most frequently asked questions and their answers (CEQ 1981). At the time of publication, CEQ stressed that the answers did not impose any additional requirements beyond those of the NEPA regulations, but rather were intended to document advice already given in other forums.

Questions 20a. and 20b. dealt specifically with worst case analyses. The question and answers were:

"20a. Q. When must a worst case analysis be included in an EIS?

A. If there are gaps in relevant information or scientific uncertainty pertaining to an agency's evaluation of significant adverse impacts on the human environment, an agency must make clear that such information is lacking or that the uncertainty exists. An agency must include a worst case analysis of the potential impacts of the proposal and an indication of the probability of their occurrence if (a) the information relevant to adverse impacts is essential to a reasoned choice among alternatives and the overall costs of obtaining the information are exorbitant, or (b) the information relevant to adverse impacts is important to the decision and the means to obtain it are not known."

"20b. Q. What is the purpose of a worst case analysis? How is it formulated and what is the scope of the analysis?

A. The purpose of the analysis is to carry out NEPA's mandate for full disclosure to the public of the potential consequences of agency decisions, and to cause agencies to consider those potential consequences when acting on the basis of scientific uncertainties or gaps in available information. The analysis is formulated on the basis of available information, using reasonable projections of the worst possible consequences of a proposed action.

For example, if there is scientific uncertainty and there are gaps in the available information concerning the numbers of juvenile fish that would be entrained in a cooling water facility, the responsible agency must disclose and consider the possibility of the loss of the commercial or sport fishery.

In addition to an analysis of a low probability/catastrophic impact event, the worst case analysis should also include a spectrum of events of higher probability but less drastic impacts."

Interestingly enough, this "guidance" probably had as great an effect on the application of RA in EISs as the regulations themselves. First, this guidance expanded on the worst case concept by introducing the notion of analysing a spectrum of events covering the range from low probability/catastrophic impact events to high probability/less dramatic events. This clearly drives the single event worst case approach towards the RA concept of analysing a range of events.

Second, the introduction of the phrase "...the analysis (worst case) is formulated on the basis of available information, using reasonable projections of the **worst possible** consequences of a proposed action", (emphasis added), opened up the legal argument that since there is no "worst possible" case, (there can always be a worse "worst case") the CEQ worst case provision was now beyond the "rule of reason" that had guided its implementation to date.

WORST CASE COURT DECISIONS

Up until 1981, there were few, if any court cases dealing with the worst provision per se. Rather, the cases dealt with the general notion of how "speculative" impact analyses should be from the standpoint of looking into the future in order to make informal forecasts about the eventual consequences of proposed actions. In general, the bottom line was that this inquiry is subject to the "rule of reason", i.e. the forecasts could be speculative but they must be reasonable.

After 1981, a number of court cases dealt directly with the worst case situation. The first major case, "Sierra Club versus Sigler" (1983) involved an EIS on a deep water port and crude oil terminal within Galveston Bay in Texas. The issue was whether a "worst case" analysis, discussing the potential impact from a catastrophic loss of the total super tanker oil cargo, was required. The Corps of Engineers (the federal agency preparing the EIS) contended that a worst case analysis was not required because of the remoteness of such a catastrophic loss within the bay. In the end, the (Appeals) court decided that the worst case oil spill was not beyond the statutory mandate of NEPA and, therefore, was required. The court went on to say:

> "...the fact that the possibility of a total cargo loss by a super tanker is remote does not obviate the requirement of a worst case analysis... The remoteness problem is instead, addressed by mandating the preparation of a worst case analysis and indicating to the decision-maker the probability or improbability of its occurrence".

Another, more far-reaching court case, "Southern Oregon Citizens Against Toxic Sprays, Inc. versus Clark" (1983), concerned the Bureau of Land Management's (BLM) proposed herbicide spraying program. It was BLM's contention that the proposed herbicide use would produce no adverse health effects. The court concluded, after determining that there was uncertainty among the scientific community regarding the carcinogenicity of the herbicide 2-4-D, that "the BLM's belief that its herbicides are safe does not relieve it from discussing the possibility that they are not, when its own experts admit that there is substantial uncertainty".

Note that the first of these cases, "Sigler", interpreted worst case as applying to a low probability-high impact situation whereas the second, interpreted worst case as applying to a case where a lack of information existed about the probability of any adverse effect occurring from the action.

Two subsequent, but related cases, "Save Our Ecosystem versus Clark" and "Merrell versus Block" (1984) interpreted the worst case provision as requiring an analysis of a spectrum of events covering a range of probabilities. In the "Merrell" case, the court determined that the Forest Service was required to perform original research on the effects of the herbicides rather than to rely on the safety information previously developed by EPA.

These cases not only broadened the concept of the worst case requirement, in fact moving it towards the RA concept, but they also raised much concern that there may be unlimited possibilities in defining a legally defensible worst case scenario.

WORST CASE REGULATORY REVISIONS

As a first attempt at clarifying the meaning of the worst case provision, CEQ proposed a Guidance Memorandum for Federal Agency NEPA Liaisons (CEQ 1983b). In this proposed guidance, CEQ introduced the notion that the worst case analysis requirement is triggered by an "initial threshold of probability" and that an agency need only examine the **reasonably foreseeable** effects of its proposed action. The proposed guidance concluded by saying that speculative information or potential adverse impacts with an extremely low probability of occurrence could not be considered "essential to a reasoned choice among alternatives". Consequently, such information or potential impacts would not meet the threshold of "reasonable foreseeability" requiring preparation of a worst case analysis.

CEQ received numerous responses to this proposed notice and produced a summary of 64 of these responses (CEQ 1983c). The concerns raised in these comments can be categorized as follows:

Federal Agencies: 6 of the 10 federal agencies believed that guidance on worst case would be helpful but that the use of the "reasonably foreseeable" standard as the triggering mechanism was inappropriate. The concern was that the low probability - high risk event would be eliminated from consideration by the proposed approach.

Environmental Groups: Most of the environmental groups' responses strongly opposed the guidance on the grounds that it would virtually eliminate the worst case requirement. The National Wildflife Federation (NWF) suggested that guidance on "how" rather than "when" would be more appropriate.

Private Citizens: Most of the private citizen responses indicated a perception that the proposed guidance would "weaken" the worst case requirement, and, therefore, mask the true risks of projects. An interesting exception to requests for keeping the worst case approach was a comment which recommended "probabilistic risk assessment" instead. This

comment suggested that risk should be measured by asking "how should society spend money to save lives. For example, society spends $25,000 dollars per life saved in our cervical cancer screening programs, and $200,000,000 dollars per life in nuclear safety programs".

State, Local Governments: The four respondents in this group all expressed similar concerns as the federal agency respondents about any change that would eliminate the low probability - catastrophic event analysis.

Business Industry: All thirty-two of these respondents supported the guidance or wanted further restrictions on the use of worst case analysis.The general theme within this group was that existing worst case scenarios have little or no basis in reality and cause undue concern about potential project impacts.

In the face of this opposition, CEQ withdrew the proposed guidance and began to consider a regulatory change to that section. After additonal interagency debate, CEQ published an amendment to their regulations in April 1986 (CEQ 1986) which stated, in part:

"Section 1502.22. Incomplete or Unavailable Information

When an agency is evaluating reasonably foreseeable significant adverse effects on the human environment in an environmental impact statement and there is incomplete or unavailable information, the agency shall always make clear that such information is lacking.

(a) If the complete information relevant to reasonably foreseeable significant adverse impacts is essentinal to a reasoned choice among alternatives and the overall costs of obtaining it are not exorbitant, the agency shall include the information in the environmental impact statement.

(b) If the information relevant to reasonably foreseeable significant adverse impacts cannot be obtained because the overall costs of obtaining it are exorbitant or the means to obtain it are not known, the agency shall include within the environmental impact statement:

1. a statement that such information is incomplete or unavailable;

2. a statement of the relevance of the incomplete or unavailable information to evaluating reasonably foreseeable significant adverse impacts on the human environment;

3. a summary of existing credible scientific evidence which is relèvant to evaluating the reasonably foreseeable significant adverse impacts on the human environment; and

4. the agency's evaluation of such impacts based upon theoretical approaches or research methods generally accepted in the scientific community.

For the purposes of this section "reasonably foreseeable" includes impacts which have catastrophic consequences, even if their probability of occurrence is low, provided that the analysis of the

impacts is supported by credible scientific evidence, is not based on pure conjecture, and is within the rule of reason."

This amendment also rescinded question #20 of the "40 most asked questions" with the promise that new guidance would be provided. In the discussions leading to this amendment, CEQ stated that:

> "...in complying with NEPA, agencies must fairly analyse and comment upon the consequences of their actions in the face of missing information in an EIS...the proposed regulation retains the duty to describe the consequences of a remote, but potentially severe impact, but grounds the duty in evaluation of scientific opinion rather than in the framework of a conjectural "worst case analysis". Section 1502.22 must, of course, be read in the context of the more general requirements for preparation of an EIS (40 C.F.R. 1502, et seq.). These include the rigorous evaluation of the direct, indirect and cumulative impacts of a proposed action, alternatives to the proposed action, and appropriate mitigation measures. (40 C.F.R. 1502 et seq.)" "...the Council has chosen to impose scientific credibility as the "threshold" to trigger the requirements of the proposed regulation. In identifying potentially significant adverse impacts, an agency must forecast those consequences which have a low probability of occurrence but have potentially catastrophic consequences when there is credible scientific support to suggest that the impact could occur as a result of the proposed action. The agency is not required to include opinions about or an evaluation of impacts which are based on pure conjecture, without a sound rationale or valid data..." (CEQ 1986).

APPLICATION OF WORST CASE RISK ASSESSMENT IN EISs

For the most part, development of the analytic methods and methodologies used in the EIS process has been left to the technical experts in the field with rather loose oversight by the federal agencies involved. With the exception of the work done by the U.S. Nuclear Regulatory Commission (1977), the utilization of RA techniques was practically nonexistent in the environmental assessment development period prior to the 1978 CEQ Regulations. After the regulations were promulgated, with its worst case provision, the incorporation of probability based analyses and the use of worst case, or extreme, conditions in analyses began to appear. However, only after the recent court cases described above has the development of formalized RA techniques began to appear in EISs. These RA applications have been oriented towards the toxic chemical/human health type impacts. The development of RA as applied to the natural environment, while given a shot in the arm by these events, has still not progressed in any uniform manner.[1]

---

[1] Paradine and Fay (1985) have done a rather extensive survey of the use of risk assessments in U.S. EISs and have documented the wide variety of approaches that have and are being used. The paper in this volume also provides a good discussion of the relationship between the worst case and the probability based risk assessments used in U.S. EISs.

In general, the current U.S. EIS "risk assessments" efforts fall into one of the following categories:

Psuedo worst case as a screening tool: This category exists because the explicit term, "worst case conditions" are used in EISs to describe it. (Adding to the confusion of using the "rule of reason" to define the worst case.) In this situation, agencies utilize extreme values from existing data bases in deterministic models to estimate a "worst case" situation (in analysis of the impacts of highways on air quality, for example, atmospheric data is selected from the "worst" time period of record) (U.S. Federal Highway Admin. 1984). This may not be a bad idea for preliminary planning purposes, but it is not a rational worst case situation since it does not account for coincidental "extreme" events (such as maximum traffic flow and adverse traffic mix).

Assumed worst case analysis as an "add-on": In these cases, the analyst simply adds a worst case scenario to the existing analysis framework. The worst case scenario generally involves an a priori assumption that a "catastrophic" release event has happened and the release event could pose a hazard to humans or the natural environment (U.S. Corps of Engineers 1984; U.S. Mineral Management Service 1984). In most cases, these results are presented in the EIS without a discussion of the uncertainties in the data and without linking the resulting exposure levels to associated levels of environmental impact.

Application of the general RA techniques: In these cases, the analysis generally deals with all four stages of the RA process (hazard assessment, dose-response development, exposure evaluation, and risk characterization) and will more or less deal with the presentational attributes of RA (discussion of assumptions, weight of evidence for qualitative judgements, etc.). In this framework, "worst case" assumptions can play a role in all four stages of analysis. The risk assessments associated with pesticide applications have to date been the best example of RA in U.S. EISs (U.S. Dept.of Agriculture 1985; U.S. Drug Enforcement Admin. 1984). However, while these assessments have analysed impacts on both humans and the natural environment, the analytical process is not as strong when applied to impacts on the natural environment. It is simply too difficult, in most cases, to develop dose-response relationships for the natural environment. The major characteristic of analyses in this category is that it results in a qualitative description of the impact on natural systems.

CONCLUSIONS

Two things seem clear from the foregoing discussion. First, while the "full disclosure" requirements of NEPA do in effect, lead to risk assessment techniques being utilized, it is the development of the standardized risk assessment framework that is furthering its application in environmental assessments. Second, while the standardized RA steps, as developed for human health impacts from toxic chemicals, is useful as a framework, the specific analytical approaches for application to the natural environment still have to be developed on a case by case basis. Given the difficulty of developing relevant dose-response information for ecosystem impacts and the time and money constraints intrinsic to the environmental assessment process, increased efforts to improve the rigour with which qualitative judgements are incorporated in the process seem essential.

REFERENCES

Council on Environmental Quality (CEQ). 1978. National environmental policy act regulations. 40 CFR Parts 1500-1508, November 29, 1978.

Council on Environmental Quality (CEQ). 1981. Forty most asked questions concerning CEQ's National Environmental Policy Act Regulations. FR 46(55).

Council on Environmental Quality (CEQ). 1983a. Guidance regarding NEPA regulations. 48 FR 34263.

Council on Environmental Quality (CEQ). 1983b. Proposed guidance memorandum for federal agency NEPA liaison. 40 CFR Parts 1502 and 1508, FR 48(156).

Council on Environmental Quality (CEQ). 1983c. Memorandum: proposed guidance on worst case analysis requirement. Dinah Bear to respondents to August 11, 1983 request for comments, November 25, 1983.

Council on Environmental Quality (CEQ). 1985. National environmental policy act regulations - Proposed Amendment to 40 CFR 1502.22, 50 FR 32234.

Council on Environmental Quality (CEQ). 1986. 40 CFR Part 1502. National environmental policy act regulations, Final rule, 51 FR 15625, April 25, 1986.

Ehrlic, A.M. 1985. Initiatives in risk assessment research. Presented at the 1985 Washington Conference on Environmental and Health Risk Assessment, October 28-29. Sponsored by Inside EPA Weekly Report, Washington, D.C.

Flamm, W.G. 1985. Update on FDA risk assessment. Presented at the 1985 Washington Conference on Environmental and Health Risk Assessment, Oct. 28-29. Sponsored by Inside EPA Weekly Report, Washington, D.C.

Paradine, P. and Fay, T. 1985. Risk assessment in American EIA. Draft Report. Federal Environmental Review Office, Hull, P.Q.

Save Our Ecosystem <u>versus</u> Clark and Merrell <u>versus</u> Block. 1984. 747 F.2d. 1240, 9th Cir. 1984.

Sierra Club <u>versus</u> Sigler. 1983. 695 F.2d. 957, 5th Cir. 1983.

Southern Oregon Citizens Against Toxic Sprays (SOCATS), Inc. <u>versus</u> Clark. 1983. 720 F.2d. 1475, 9th Cir. 1983.

United States Corps of Engineers. 1984. Georgetown refinery, Georgetown, South Carolina. Draft EIS.

United States Department of Agriculture (Forest Service). 1985. Gypsy moth suppression and eradication projects. Draft Addendum to the final EIS as supplemented-1985.

United States Drug Enforcement Administration. 1984. Eradication of Cannabis on federal lands and intermingled forest and rangelands in the continental U.S. Draft ES.

United States Environmental Protection Agency (U.S. EPA). 1984. Risk assessment and risk management - framework for decision-making. EPA 600/9-85-002.

United States Environmental Protection Agency (U.S. EPA). 1986. Guidelines for carcinogen risk assessment, guidelines for mutagenicity risk assessment, guidelines for the health risk assessment of chemical mixtures, guidelines for the health assessment of supect developmental toxicants, and guidelines for exposure assessment. U.S. Federal Register, 51(185).

United State Federal Highway Administration. 1984. Fundamentals of air quality for highway planning and project development. U.S. Department of Transportation, Washington, D.C.

United States Mineral Management Service, DOI. 1984. Oil and gas lease sale - Diaper Field. Draft EIS.

United States National Research Council (U.S. NRC). 1983. Risk assessment in the federal government: managing the process. National Academy Press, Washington, D.C.

United State Nuclear Regulatory Commission. 1977. Calculation of annual doses to man from routine releases of reactor effluents for the purpose of evaluating compliance with CFR Part 50, Appendix I. Regulators Guide 1-109. U.S. NRC, Washington D.C.

United States Office of Science and Technology Policy. 1985. Chemical carcinogens: a review of the science and its associated principles. U.S. Federal Register, 50(50).

CHAPTER 4
RISK IDENTIFICATION AND ESTIMATION

James E. Dooley,
Faculty of Management, University of Toronto

INTRODUCTION

The purpose of this chapter is to survey the principles and some techniques for analysing hazardous situations from a technical or scientific perspective.It complements the social and political perspectives presented in other chapters. Ideally, technical analysis should provide objective and unbiased information on risks, using the methods and data that are available or can be developed within practical limits. However, many factors may bias the choice of technique and data. The checks on the quality of the analysis are public scrutiny and peer review.

This chapter focusses on the systems that need to be reviewed under the federal and provincial legislation on environmental impact assessment (EIA). For example, the federal Environmental Assessment and Review Process (EARP) applies to those projects, programs and activities "undertaken or sponsored by federal departments or agencies, those for which federal funds are solicited and those involving federal property" (Couch 1983). In practice, as Paradine notes in this volume, the provisions to date have been applied mostly to major energy projects such as the Beaufort Sea Hydrocarbon Development Proposal, the Alaska Highway Gas Pipeline, and the Lower Churchill Hydroelectric Project.

One of the issues that will be discussed is the large number of disciplines that normally have to be applied to estimate and evaluate risk. Depending on the project, it may be necessary to draw upon the knowledge of engineers, statisticians, hydrologists, meteorologists, biologists, and various medical disciplines. Within the limits of one chapter, it will be possible to include only a sampling of the more important technical approaches and even those can be discussed only in a preliminary way. Sources indicate where more detailed treatment can be found.

The chapter begins with a discussion of the meaning of risk and the recognition that several definitions are possible. It is argued that a comprehensive description of risk includes the probability distribution of the possible consequences. From this several simpler expressions are possible. A general model of risk analysis is presented that provides a framework for doing the detailed analysis to arrive at the estimated probability distribution. The general model divides the analysis up into a number of convenient stages that parallel the physical process from the source of risk through to the consequences. The treatment is limited but serves to show what is involved in a detailed analysis and some of the disciplines needed. The level of detail is greater in the section on release analysis than in the next two dealing with exposure and consequence analysis.

DEFINING RISK FOR TECHNICAL ANALYSIS

The term "risk" embodies two main components. One is that a situation, action or system exists that has the potential for unwanted consequences. The other is that there exists uncertainty in some respect about the occurrence of the consequences. The uncertainty may be about where, when or whether the consequence will occur, who or what will be affected and the magnitude. This descriptive definition is adequate for general use but we need to go further to express the risk quantitatively.

Quantitative risk analysis uses two further concepts: probability distributions for the consequences and summary measures of the distributions (e.g. expected value and variance). The uncertainty about the consequence is described most comprehensively in the form of a probability distribution for the consequence. The probability distribution may be for joint consequences, or conditional on other events. Finally, risk is "defined" in particular contexts by selecting a summary measure of the distribution that:

(a) simplifies the description (contains less information than the distribution but is simpler to use); and
(b) focusses on the particular aspect of uncertainty that is important in the context.

To illustrate the idea of a probability distribution and its summaries, consider a situation, in which there could be zero or one or two and so on up to six people killed in a year of operation of a facility. Suppose the probability of each of these possibilities is as given in the table below.

Index	Consequence, number of deaths	Probability of consequence	Cumulative probability
i	y_i	$f(y_i)$	$F(y_i)$
1	0	0.14	0.14
2	1	0.27	0.41
3	2	0.27	0.68
4	3	0.18	0.86
5	4	0.09	0.95
6	5	0.04	0.99
7	6	0.01	1.00

The list of possible consequences and the probability of each is known as the probability distribution. In this case it is discrete because no fractional consequences are possible. What this shows is that there is a fairly high probability of few deaths occurring and a smaller probability of the higher numbers. In this illustration there are only a few possibilities so the probability distribution is not too extensive. But in some cases there could be a very large number of possibilities and so it is useful to summarize the information contained in the probability distribution. A number of common summaries are used.

To describe the summaries, the consequence, death, will be called "Y" and the particular number of deaths denoted by "y". The probability of y will be denoted by f(y). Using this convention the probability of three deaths will be denoted by f(3) and the probability is 0.18. The notation F(y) is called the distribution function and denotes the probability that the consequence will be less than or equal to y. For example, F(3) is equal to 0.86.

EXPECTED VALUE

This is the average, also called the mean, and is obtained by multiplying the consequences by their respective probabilities and taking the sum. Using the above notation the mean is defined by:

$$\text{mean of } Y = E(Y) = \sum_{i=1}^{7} y_i f(y_i)$$

Applying this formula to the example gives a mean of about 2 deaths per year. The mean is often used as the measure of risk.

VARIANCE

The variance gives a measure of the spread or dispersal of the possible consequences. The variance is obtained by:

$$\text{var}(Y) = \sum_{i=1}^{7} (y_i - E(Y))^2 f(y_i)$$

The variance in the example is about 1.9, in this case almost the same as the mean but that is not always so. A related measure is the standard deviation which is the positive square root of the variance. In the example the standard deviation is about 0.95 deaths.

RANGE

A simple measure of dispersion is the range. This just states the range of possible consequences. It is obtained by subtracting the minimum from the maximum consequence. In the example the range is 6-0 or 6.

PERCENTILE

Often it is desired to find the level of consequence for which the comulative probability is p. Using the notation (and the discrete case), this can be stated as: find y_i such that $F(y_i) \leqslant p < F(y_i + 1)$. The level of consequence so found is called the percentile. Usually the probability is converted to a percent which is done by multiplying by 100. Suppose the probability desired is 0.95 so the percent is 95. The level of consequence such that there is a 95 percent chance that the consequence will be less than or equal to that consequence is 4 deaths in our example.This means that in only 5 percent of the years is the number of deaths expected to exceed 4.

EXCEEDING A LEVEL OF CONSEQUENCE

A measure analogous to the percentile is the excedence probability. In this case a consequence is specified and the probability of exceeding that consequence is used as the measure of risk. Using the example it may be desired to know the probability that more than one death will occur in a year. From the above table it is found that the probability of more than one death is 0.59. Equivalently the probability of one or less death in a year is 0.41.

RETURN PERIOD

This measure is used commonly in connection with flooding. A number of years is specified. For this number of years a river crest level is determined such that on average the crest does not exceed this more often than the return period. The usefulness of this measure is that it says how often on average a given hazard level will occur. The measure is useful for planning land use on flood plains. The idea can be adapted for the example. The probability of one or less deaths in any year is 0.41 so that on average more than one death will occur every 1/0.41 or 2.43 years.

DISCUSSION OF SUMMARIES

The mean gives an "expected value" but no indication of how large or small the consequences could be. In the example the mean or average is about 2 but the consequences could be as low as zero to as high as 6 deaths. The nuclear industry is a very dramatic example in which there could be no consequence if there is no accident to the other extreme of very high consequences if an accident should occur. But in the nuclear industry the expected value would be small because of the low probabilities of severe accidents. Thus the mean alone does not give a complete picture.

When the mean is used it should be accompanied by a measure of dispersion. The standard deviation is a common measure used in statistics but this still does not give the full range of possibilities. In the example, the standard deviation is 0.95 and the range is 6. In the risk field the public is often concerned with the extreme consequence or the worst possible case.

The percentile is sometimes used to set standards for toxic substances in the water. This will be illustrated in the section on the dose response curve. This gives the probability that a person from the population exposed to a toxic substance will suffer a consequence, such as cancer, as a function of dose. Suppose that one in one million is considered a tolerable risk. That is equivalent to the 0.0001th percentile of the susceptability of the population. It remains to find the dose corresponding to this percentile which then becomes the limit for the substance in drinking water or food etc.

The probability of exceeding a given level of consequence is the converse of percentile. This would provide probability information for a specified consequence.

CONSEQUENCES

There could be a large number of types of consequence for a particular system but in practice the risk is normally expressed in terms of a few of the more important ones. In the Wash 1400 study of nuclear power (U.S. Nuclear Regulatory Commission 1975), for example, the risk was expressed in terms of two main consequences: immediate death and delayed cancer. Table 1 lists a number of categories of consequences that should be considered. These are first level consequences only and many could lead to further consequences. If, for example, the residential water supply receives a toxic substance, human health might subsequently be affected. These possibilities should be explored in the analysis.

Table 1. Consequences by receptor category

A. Consequences to Humans
- Immediate death (usually defined as death within 5 days of the event)
- Recoverable injury
- Permanent loss of some function
- Delayed (chronic) illness
- Susceptibility to a conjoint illness

B. Consequences to the Natural Environment
- Change in life support potential of air, soil, water
- Change in physical shape or topography
- Change in health or habitat of species

C. Consequences for Valued Material Objects
- Loss of an economic resource (including cost to restore)
- Loss of an historic or scenic site

GENERAL MODEL OF RISK ANALYSIS

The technical analysis of risk has several uses in EIA. One is to estimate the probability that consequences of various kinds will occur under certain design and operating conditions for a given project. This information could be used to decide whether to approve the project or not. The analysis can also be used to identify weaknesses in the design or operating procedures so that these may be improved where necessary.

The scope and role of technical risk analysis may be made clearer by using a schema of the physical process. Assume that a system exists that has the potential to lead to unwanted consequences, e.g. a liquified petroleum gas (LPG) terminal, a radioactive material disposal site, a hydro-electric dam, a hazardous waste disposal site, emissions to water or air, a pesticide application program, acid rain, a recreational activity, or a volcano. Each of these has the potential for a number of unwanted consequences; nothing beyond the planned or tolerated consequences occur as long as the plans are sound and the projects operate according to plan. We "tolerate", for instance, in the order of 4000 traffic deaths per year in Canada although we take steps to reduce this level. We also tolerate levels of acid rain at least within a program to reduce levels. However, there are surprises in some cases such as occurred as a result of using DDT, or unplanned releases such as the Seveso and Chernobyl accidents.

The problem arises when an upset to normal function occurs and the hazardous potential is released. At that point a hazardous state exists. To illustrate this, consider an LPG facility. It has the potential to explode and burn causing deaths, injuries, and environmental damage. As long as the plant operates according to plan, none of these consequences will occur. But if, for example, a storage tank should rupture, then a vapour cloud would form and a hazardous state would exist. As long as the vapour cloud does not ignite or, if it does, there are no people or property nearby, then there is no consequence except for a loss to the owner. In the example, the vapour cloud must ignite and explode at a location with people, property or valued objects (the receptors). Finally, the consequence may be realized by all or some of the people or property exposed.

There are management measures that may be taken to limit risk. We can think broadly of risk management as the planning and implementation of measures to control or reduce the risk at all stages of the physical process from release to consequences. For example, the LPG terminal may be located away from a population centre so that if an explosion occurs, fewer people will be at risk. Engineering design (e.g. tank design, relief valves, containment for released material and so on) is another important risk management option.

If enough historical data were available - which is not usually the case - we could use frequency data to estimate the probable number of people killed in LPG explosions per operating year or the probability of safety valve failure. When sufficient historical data are not available, other means of estimation have to be used. One approach is to do a detailed analysis starting with the events that could initiate the hazardous situation (e.g. a propane truck in an accident) and following through to possible consequences.

The detailed analysis can be organized using a framework that identifies the stages of the analysis. The framework given here is an extension of the one developed by Kates <u>et</u> <u>al</u>. (1985). The extensions expand the framework in the area of risk management options and the organization of a risk analysis. Figure 1 shows the process broken down into a number of stages from human and natural activities to the eventual consequences. For each stage the progression of the hazard is identified starting with the hazard potential and ending with consequences or hazard realized.

The model is divided into a series of physical processes each resulting in a risk state. The main physical processes are:

Activities:
  The human activities and natural events that create a hazard potential.

Release process:
  The episodic or continuous release of energy or material.

Exposure process:
  The transformation of energy and material in state, time or space and exposure by receptors.

Consequence process:
  The effect of energy or material on receptors.

Human and natural activities — Hazard potential — Discrete or continuous release of energy and material — Hazardous situation — Transformation of energy and material — Exposure — Effect on receptors — Consequences — Evaluation

	ACTIVITY ANALYSIS	RELEASE ANALYSIS	EXPOSURE ANALYSIS	CONSEQUENCE ANALYSIS	RISK/BENEFIT ANALYSIS
	Human and natural activities	Discrete or continuous release of energy and material	Transformation of energy and material	Effect on receptors	Evaluation
MAJOR UNCERTAINTY	Needs Human/natural interactions Future needs Future population and events	When, Where How much release	Location Concentrations Recipients	Magnitude of effect Who/what affected	Consequence magnitude: Human Natural environment Material Attitudes and value system Political
TYPICAL DISCIPLINES	Economics Physical and biological sciences	Earth sciences Engineering Statistics	Engineering Physical, medical and biological sciences Geography	Medicine Biology Epidemiology Toxicology	Economics Social sciences Ecology Political science Public health
SOME MODELS AND METHODS	Demographic Geophysical Forecasting	Fault tree analysis Reliability models Probability and statistical models	Dispersion in air, ground and surface water Food chain models	Dose threshold response	Economic evaluation (cost/benefit analysis) Decision theory Utility theory Multi-criteria models
SOME MANAGEMENT OPTIONS	Technology Location	Design details Operating procedures Maintenance procedures	Avoidance Containment Neutralization	Neutralization Recovery	Decision procedure Evaluation procedures Compensation

Fig. 1. A framework for risk analysis and management
(Adapted from Clarke University Hazard Assessment Group 1983)

Within each of these there are many more detailed processes but this division is sufficient for present purposes. The schema in Fig. 1 identifies the progression of the system from potential to consequence. Each of the physical processes requires a different form of analysis. These are described below. The evaluation stage, including economic, political and social considerations, is discussed elsewhere.

HUMAN AND NATURAL ACTIVITIES (ACTIVITY ANALYSIS)

As noted above, all activities create risk. However, some are so small that we do not worry about them. Others, such as volcanic eruptions, cannot be controlled so we accept their existence but may try to mitigate their consequences. Still others have significant risk, such as the use of automobiles, but we choose to accept the activity to enjoy the benefit. Activity analysis helps us to decide whether or not to proceed with projects, programs and activities over which we have control; it can be thought of as screening to select the options to be considered in more detail. In many activities, material is removed rather than added and this may also have hazard potential (e.g. removal of trees leading to flooding).

EPISODIC OR CONTINUOUS RELEASE OF ENERGY OR MATERIAL (RELEASE ANALYSIS)

In this stage, analysis focusses on the events and pathways leading to a release of energy or material. There are two main categories. In the first, the process is designed to operate in a way that, if nothing goes wrong, there will be no harmful consequences. When something does go wrong there is an unplanned release of material or energy. Consider, for example, the transport of liquid chlorine by truck. A series of events might occur such as: ice forms on the road, the truck slides, a collision results, and the tank ruptures, releasing chlorine. The series of events changes an activity with hazard potential into a hazardous situation. The techniques for analysis at this stage include fault tree analysis, event tree analysis and statistical models. Fault tree analysis is discussed in more detail below.

A second category is where the material or energy release is planned. Sulphur dioxide emission from a power generating station is an example. The amount may vary over time and is not regarded as unusual unless planned limits of release are exceeded. Even though planned, the presence of the material may create a hazardous situation and may lead to unwanted consequences under normal operation. Since the releases are planned there is no special technique in release analysis for examining them; they are part of the activity description. The analysis of their effect or consequences will be analysed in later stages. The possibility of episodic release can be analysed using the same techniques as for unplanned releases. Risk is managed at this stage largely by engineering design, maintenance and operating procedures, when dealing with man-made systems.

TRANSFORMATION OF MATERIAL OR ENERGY (EXPOSURE ANALYSIS)

The released material or energy creates a hazard potential. If this should disperse, transform to something harmless, get diluted or move to a location where there are no receptors, then the chain is broken and no consequences will result. Exposure analysis examines the ways that receptors become exposed and the intensity and frequency of exposures. Exposure changes the risk state from one of hazard potential to "hazard received" or "receptors exposed".

The transport of chlorine by road will illustrate this stage. In our example, the hazardous situation is created when chlorine is released from the tank truck as the result of an accident. Now there is a cloud of chlorine gas, which is heavier than air, uncontained at ground level. The chlorine gas around the truck likely will be at concentrations lethal to humans and animals. The analyst would like to know how the chlorine cloud is most likely to disperse in relation to human and animal populations. To analyse this situation, we use microclimatic data and dispersion models as well as information on the kinds and locations of receptors.

As a second example, consider the release of propane from a storage tank. If the cloud ignites we are concerned with the shape and location of the flame front and the energy released. The population and materials in this zone are the potential receptors so we must consider who or what is present in the exposure zone.

Both of the above examples are episodes creating risks. An example of planned release is sulphur dioxide and oxides of nitrogen which lead to acid rain. The exposure analysis requires models to analyse the transformation of sulphur dioxide to sulphuric acid and oxides of nitrogen to nitric acid. Then models of dispersion and transportation are used to trace the amounts and location of the acid in the atmosphere. Several pathways of exposure may be distinguished. One is through the respiratory system of humans and and other animals, another is by deposition into lakes, soil, and on buildings. Thus acid rain can create a hazard for humans, the natural environment and valued material objects.

EFFECT ON RECEPTORS (CONSEQUENCE ANALYSIS)

By this stage the receptors have been exposed to the hazardous material and energy at various levels of intensity. The uncertainty is now about the consequences that could result.

There are several models and analytical tools to estimate consequences to human health, but the two main lines of investigation are epidemiology and toxicology. The first uses observations on what has happened in similar circumstances in the past. By monitoring, case studies, and through statistical analysis, we can get some understanding of what consequences have been realized in the past and can try to use these as a basis for predicting future consequences. The second approach is to attempt to extrapolate from experiments, typically on mammals, to predict consequences where the hazardous substance may be toxic, carcinogenic, mutagenic, or teratogenic at various levels of concentration and exposure.

In this section we have described how the physical process can be broken into convenient stages for analysis and that each stage requires different forms of analysis and often involves different disciplines. There are different management options to reduce risk at the various stages. How are the analyses put together for a risk analysis? We can deal with the question in two stages: risk identification and risk estimation. Risk identification, in effect, must trace the pathways through the whole process to see what possible consequences might result and what pathways need to be considered. In a second pass the probabilities of the various consequences are estimated. This may be done directly by estimating the probabilities of consequences from historical data (e.g. motor vehicle accidents) or one may proceed through a detailed study of the pathways, or sequences of events, and assign and combine the probabilities. This topic is taken up again in more detail below.

The evaluation stage of risk analysis is shown as the last column of Fig. 1. As the last step we need an evaluation of the project in total to decide whether or not the risk is acceptable. So much attention has been paid to the final decision that it tends to obscure the numerous decisions with risk implications that are made at the many stages as the project proceeds. In an EIA, the many decisions, from project conception to final operation and maintenance, including emergency preparedness and decommissioning, should be evaluated from the risk persepctive. A simple but obvious example would be the evaluation of options on where to locate a toxic chemical plant in the vicinity of a population centre. One option, probably the best, would be to locate it on the leeward side of the population centre with respect to the prevailing winds so that, if there should be a release, the probability of exposure would be minimized. This example also illustrates the kinds of risk management options that are considered at the activity stage of the framework.

RISK IDENTIFICATION

Risk identification is the examination of human and natural processes and activities to identify actual or potential consequences for humans, property, and the natural environment. Let us take as an example an EIA of siting a hazardous facility. Risk can be identified by examining the physical processes in the project, through the hazardous situation (a release has occurred), the exposure mechanism (receptors have been exposed) to the consequences. A material balance can be used to indicate the potential release of a hazardous substance but further examination is needed to identify the fate of the material, if released, and the consequences that could result.

A consequence is realized when events or processes occur along the path described in Fig. 1. The checklists described below show that risks can be classified by several dimensions (descriptors). Several of these pertain to different places along the path so the checklists should be used in conjunction with a specific configuration of Fig. 1 applicable to the EIA of a particular project or program.

CHECKLISTS

Checklists have been used extensively in environmental impact analysis and are a useful starting point to identify risks. For practical purposes, hazards or risks can be treated as those impacts that are uncertain in some respect (e.g. when, where, who, magnitude). The impacts with "important uncertainty" are candidates for risk analysis. A risk has important uncertainty if it is a critical feature of the situation. For instance, the uncertainty about the amount of toxic release from a chemical plant is important because the consequences of a release may be very serious.

Several attempts have been made to create taxonomies of risk. These cover a spectrum of risks well beyond those of concern to EIA. One approach to classifying (and comparing) risks is to list a number of situations and to attach descriptors to each. The descriptors explain the risk and give information on its magnitude. The Clark University Hazard Assessment Group (1983) has developed a list of twelve descriptors for characterizing the level of hazard at each of the stages of hazard progression as set out in Fig. 1. The descriptors are listed in Table 2 together with suggested measures to express the magnitude of the hazard for each descriptor as an index: an integer between 1 (low level) and 9 (high level). The description of the measures for the first two and the last three descriptors are given in Table 2 as examples. Although the measures suggested are arbitrary, they are useful to compare risks. A particular hazard is characterized using a twelve digit code, one digit for each of the descriptors. Suppose the following is the assessment of the hazard for the first two descriptors for an LNG explosion hazard; it is not intended to harm living organisms and has a spatial extent of 100,000 m^2. Using the formulae of Table 2 these descriptors would be denoted by the two digits 36. This scheme provides a common basis for comparing risks for each characteristic separately.

HAZARD POTENTIAL MATERIAL FLOW AND BALANCE

Other useful devices for identifying risks are the material balance and flow diagrams used to study flow processes. These identify the materials and operations, particularly in chemical processes, from the input materials to the outputs. The materials are identified by composition and physical conditions such as flow rates, pressure and temperature. Also the operating conditions of the processes are identified. In risk analysis this device serves the further purpose of identifying potential hazards. In EIA we could use the material balance and flow diagram to identify effluents that may or may not be harmful. If, for instance, the process has a large volume of concentrated chemicals under pressure then this should be identified as a risk to be estimated and evaluated. The presence of the material would not cause harm under planned containment. The uncertainty that makes this a risk is the possibility that it may be released accidentally. In general, by following through the material flow and balance, the risk analyst should be able to identify the potential hazards and have some idea of their possible magnitude.

Table 2. Hazard descriptor definitions

TECHNOLOGY DESCRIPTOR

1. **Intentionality** Measure the degree to which technology is intended to harm using a categorical scale; 3 - not intended to harm living organisms; 6 - intended to harm non-human living organisms; 9 - intended to harm humans.

RELEASE DESCRIPTORS

2. **Spatial Extent** (S) Measure the maximum distance over which a single event has significant impact, using a logarithmic scale, $1<s<9$, where $s = \log_{10}d + 1$ rounded to the nearest positive integer, and d is the distance in meters.

3. **Concentration** Measure the concentration of released energy or materials.

4. **Persistence** Measures the time over which a release remains a significant threat to humans.

5. **Recurrence** Measures the mean time interval between releases above a minimum significant level.

EXPOSURE DESCRIPTORS

6. **Population at risk** Measures the number of people potentially exposed to the hazard.

7. **Delay** Measures the delay time between exposure to the hazard release and the occurrence of consequences.

CONSEQUENCE DESCRIPTORS

8. **Human mortality** (annual) Measures average annual deaths in the US to the hazard in question.

9. **Human mortality** (maximum) Measures the maximum credible number of deaths in a single event.

10. **Transgenerational** Measures the number of future generations which are at risk for the hazard in question, using a categorical scale; 3 - hazard affects the exposed generation only; 6 - hazard affects children of the exposed generation, no others; 9 - hazard affects more than one future generation.

11. **Nonhuman mortality** (potential)
Measures the maximum potentional nonhuman mortality, using a categorical scale; 3 - no potential nonhuman mortality; 6 - significant potential nonhuman mortality; 9 - potential or experienced species extinction.

12. **Nonhuman mortality** (experienced) Measures nonhuman mortality that has actually been experienced on a categorical scale; 3 - no experienced nonhuman mortality; 6 - significant experienced nonhuman mortality; 9 - experienced species extinction.

Source: Clark University Hazard Assessment Group 1983

RISK ESTIMATION (RELEASE ANALYSIS)

In this section we begin with the assumption that a system is built or operated that has the potential to cause a hazardous situation. The problem is to analyse the system to determine the ways that material or energy could be released, the possible amounts, and their probability. The problem is primarily an engineering concern where we need to know about system reliability particularly for man-made systems. For natural hazards such as volcanic eruptions, hurricanes, or floods, we need information from the natural sciences, geology and meteorology, for instance.

Two methods are widely used for risk analysis of engineering systems: fault tree analysis and event tree analysis. Both of these methods were developed because of the need for careful and detailed analysis of possible failure in aircraft and nuclear power plants. It is costly to make a detailed analysis but the possible consequences of system failure in many cases more than justify the cost. The analysis will often show where design changes could improve overall safety.

FAULT TREE ANALYSIS

Fault tree analysis starts with the hazardous state (or event) and works backward through the system to identify all of the events or component failures that could lead to the hazardous state. This is more efficient than examining all possible initiating events, many of which do not lead to hazardous events. For an extensive treatment of fault tree analysis see Henley and Kumamato (1981), Veseley et al. (1981) and McCormick (1981).

A simple example is constructed here to illustrate the method of fault tree analysis. In this section we consider the probability that an automobile is failing to stop. The example is chosen because of its widespread familiarity, and it takes human factors into account explicitly. Most practical applications are much more complicated than this.

Figure 2 shows the events to be considered in the example. The probability of occurrence for components failure and human error is given in Table 3. The vehicle fails to stop if the main braking system fails to work and the driver does not have the presence of mind to use the hand brake when the brakes fail (a human factor), or if the hand brake is tried but does not work. Using the symbols shown on Fig. 3, the vehicle is unable to stop if events E_1 or E_2 occur. As we go into more detail, the main brakes fail if the brake lever is broken, C_2, or if there is a loss of fluid E_5. Going further, the loss of fluid may be the result of a cylinder leak C_5 or a line rupture C_6. In some cases either one or the other of two events can lead to failure (C_5 or C_6) whereas in other cases more than one event is needed for failure (E_3 and E_4). Moving to the events under E_2 a total brake failure occurs if both the handbrake and normal brakes fail. The event E_3, brakes fail are explored under the branch below E_1. The handbrake fails if the cable is broken, C_3, or if it is not adjusted, C_4.

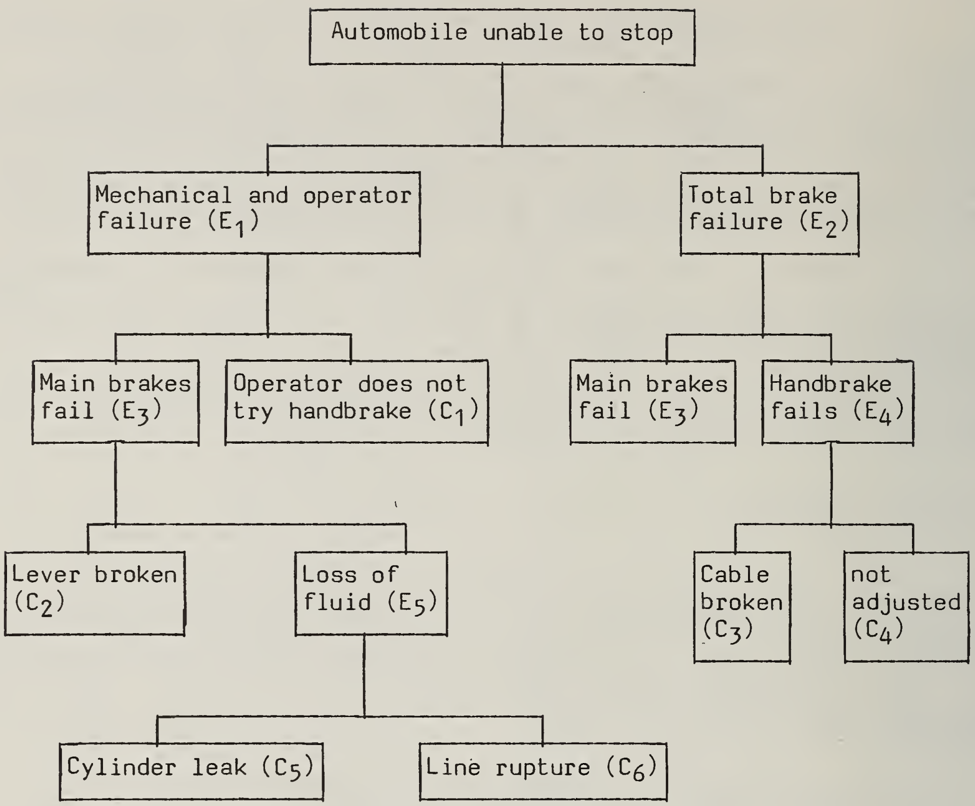

Fig. 2. Event leading to failure of automobile to stop

In the example we have used words such as "total brake failure, if hand brake fails or if brakes fail" to discuss combinations of events. The description of such combinations and their manipulation is facilitated using a branch of mathematics known as Boolean algebra, sometimes called the algebra of sets. Symbols are used by convention to represent ways that sets are combined. The fault tree diagram shows which intermediate events (Es) can lead to the hazardous event (A). The component failures that cause the intermediate events (or hazardous event), are shown at the bottom of the tree (Cs). Gate symbols are used to represent "and" and "or" as follows:

 **AND** - Output fault occurs if all of the input faults occur

 **OR** - Output fault occurs if at least one of the input faults occurs

The events in Fig. 2 are represented showing these rules of combination in Fig. 3, which is known as a fault tree.

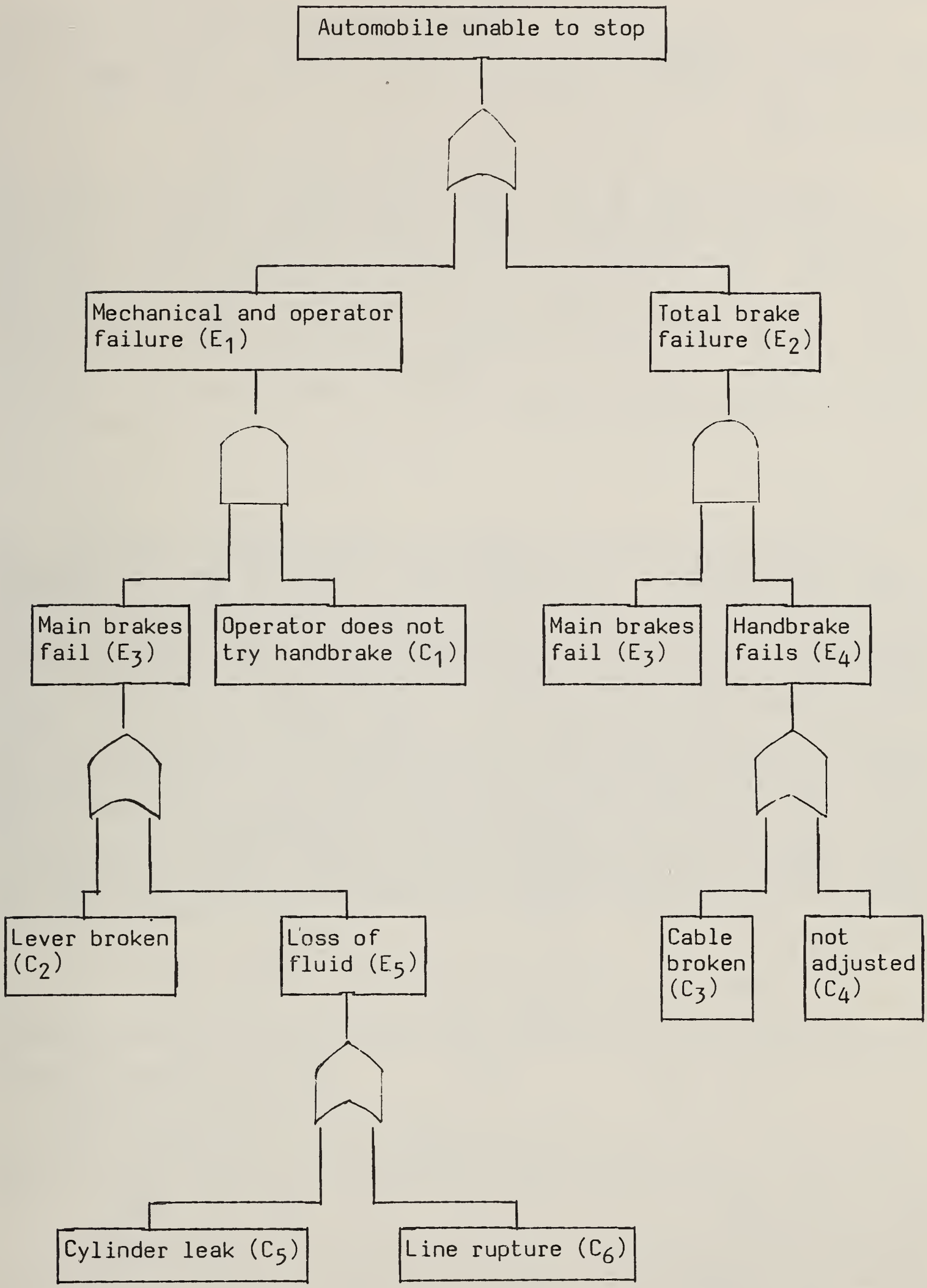

Fig. 3. Events leading to failure to stop

MINIMAL CUT SETS

It is desired to know the probability that the hazardous event will occur. Before considering the probabilities, however, we need to know the paths from the bottom to the top of the tree that could lead to the hazardous event. One way of finding these is to use the notion of a minimal cut set.

A minimal cut set is a group of events such that all must occur for the hazardous event to occur. In an engineering context the events typically would be component failures leading, through a series of events, to a release. In the example, for instance, one minimal cut set is C_5 and C_3 (Fig. 3). That is, if there is a loss of fluid due to a cylinder leak the main brakes fail and, if the cable is broken, the hand brake fails so the car fails to stop. There are several minimal cut sets in the example. These are shown in Table 4. For a formal presentation on how to find all of the minimal cut sets and the probability of occurrence the reader is referred to Veseley *et al*. (1981).

PROBABILITIES

Risk may be expressed quantitatively using probability methods. If, in the above example, the probability of each of the six component failures is available (Table 3) and they are assumed to be independent, the probability of each minimal cut set could be found by multiplying the probabilities. This would give the probability of the hazardous situation a particular combination of component failures, the combination in the minimal cut set. If the minimal cut sets were mutually exclusive (i.e. no two contain common components, e.g. $C_1.C_2$ and $C_3.C_4$ have no common components) then we could find the probability of the hazardous event, failure to stop, by adding the minimal cut set probabilities. If they are not mutually exclusive, then adding them overstates the probabilities. However, overstating the probabilities is conservative, so they are usually added. If the probabilities were large, the sum of minimum cut probabilities could exceed one which would violate an axiom of probability theory but since they are usually small, the sum is normally very much less than one.

Table 3. Component failure probabilities[1]

Component or human factor	Symbol	Failure rate per 250 hours of operation or probability of unavailability
Handbrake not tried	C_1	1.2×10^{-3}
Brake lever broken	C_2	1×10^{-7}
Handbrake cable broken	C_3	1.5×10^{-5}
Handbrake not adjusted	C_4	4×10^{-2}
Cylinder leaks all fluid	C_5	1.3×10^{-6}
Brakeline rupture	C_6	1.5×10^{-6}

[1]The probabilities were fabricated for the example only and must not be used in real situations

Table 4. Minimal cut set probabilities

Minimal cut set	Component probabilities (multiplied)	Cut set probability ($\times 10^{-9}$)	Percent of total probability
$C_5 \cdot C_1$	$(1.3 \times 10^{-6})\ (1.2 \times 10^{-3})$	1.56	1
$C_6 \cdot C_1$	$(1.5 \times 10^{-6})\ (1.2 \times 10^{-3})$	1.8	2
$C_2 \cdot C_1$	$(1 \times 10^{-7})\ (1.2 \times 10^{-3})$	0.12	-
$C_5 \cdot C_3$	$(1.3 \times 10^{-6})\ (1.5 \times 10^{-5})$	0.0195	-
$C_5 \cdot C_4$	$(1.3 \times 10^{-6})\ (4 \times 10^{-2})$	52.	44
$C_6 \cdot C_3$	$(1.5 \times 10^{-6})\ (1.5 \times 10^{-5})$	0.0225	-
$C_6 \cdot C_4$	$(1.5 \times 10^{-6})\ (4 \times 10^{-2})$	60.	50
$C_2 \cdot C_3$	$(1 \times 10^{-7})\ (1.5 \times 10^{-5})$	0.0015	-
$C_2 \cdot C_4$	$(1 \times 10^{-7})\ (4 \times 10^{-2})$	4	3
Sum of probabilities of minimum cut set		119.5×10^{-9}	100

There are several ways to express component failure probability. One is to state the failure rate per hour or year of operation. This is sometimes referred to as the component failure rate. A second kind of failure is "unavailability" when called upon and is also expressed as a probability. There are several published sources of data on component failure probabilities. Henley and Kumamoto (1981) devote a chapter to a discussion of data bases commenting on sources and reliability, and give examples of failure rates. These can be used to find the probability (failure rates) of each minimal cut set. If the failure rates are not available from published sources, then other sources need to be found or judgements made. In the automobile example, accident data could possibly be used to estimate component failure (unavailability). For example, the frequency of line rupture might be estimated as failures per vehicle mile by taking the ratio of failure per year (from accident reports) divided by the vehicle miles driven per year. In practice, of course, a more precise analysis would be needed.

Theoretical ways of obtaining distributions of time to failure are given in Henley and Kumamoto (1981) and McCormick (1981). One model assumes that the failure rate is constant so the probability of failing is the same for all intervals of time but the cumulative probability of failure increases with time.

We can now return to the simple example. Suppose for illustrative purposes, the failure rates for components and human error are as given in Table 3. Note that these are strictly fabricated numbers. The component failure probabilities are used to find the probability for each minimal cut set as shown in Table 4. The last column of Table 4 shows the minimal cut set probability as a proportion of the total probability. This gives a measure of the relative importance of each minimal cut set and indicates the combinations of components that should be improved (or protected or compensated) in some way if risk is to be reduced. In the example, $C_5 \cdot C_4$ and $C_6 \cdot C_4$ are the important ones. So to reduce risk, note that C_4, handbrake not adjusted, has the largest probability in each combination and so improve this component.

The discussion of fault tree analysis has been simplified. It has been included, however, to indicate the basis for estimation of release (energy in this example) and some of the sources, and to show that it is not a trivial matter to estimate risk. These analyses are applicable in EIA, e.g. in estimating probabilities of oil release from offshore drilling (cf. Paradine, this volume).

EXPOSURE ANALYSIS

The aim of this section is to indicate the pathways by which the hazard reaches the receptors. This will show clearly that there are many analyses to choose from depending on the nature of the hazard. Thus in risk estimation a major problem is to identify the nature of the exposure process and to select an appropriate model or procedure from those available to estimate the exposure intensity, with probabilities.

Exposure requires the presence of a hazardous medium and the presence of sensitive receptors. Exposure analysis is concerned with how the released hazard transforms and the pathways by which it reaches the receptors. The exposure process could be called simple if the receptor and the hazard come together in a direct way, e.g. when people (receptors) are caught in flood waters (hazard). In other cases the pathway is more complex (e.g. acid rain from sulphur dioxide). First consider the transformation of state. Sulphur dioxide (SO_2) released to the atmosphere is converted to sulphur trioxide (SO_3) in an oxidation process. Then the sulphur trioxide is converted by hydrolization to form sulphuric acid. There is movement in space during conversion and there is also a transformation in time because it takes released sulphur dioxide some time to go through the change of state and eventually to be deposited on the earth's surface. To follow the exposure process further we have to consider the receptors. There are receptors in each of the three categories mentioned earlier: humans, the natural environment, and property. Our intent is not to follow all of the pathways or to list all of the receptors but to use the example to illustrate the possible complexity of the exposure process. Consider first, valued material objects such as buildings. The buildings receive acid directly with the rain so that pathway is fairly simple. Humans breathe the air and so are exposed to acid through the respiratory system, also a fairly direct exposure. Fish in lakes are also receptors but here the pathway is more complex. Some acid is deposited directly into the lake as rainfall. Acid also falls on the land and leaches other elements (e.g. aluminum) from the soil into the lake where they are ingested by fish.

The consequences of these exposures will not be considered until the next section on consequence analysis. Nevertheless, it is necessary to look ahead to the possible consequences to decide which pathways it is practical to follow because there are often many possible ones. The identification and selection of receptors also needs to be considered in some detail. At a meso level the receptor may be taken to be human. The toxicologist will often want to carry that further and consider particular organs as receptors. Also the toxicologist will include in the exposure analysis the means of entry to the body which could be through ingestion, respiration or through the skin.

The main pathways for exposure are through the air, soil, groundwater, and surface water. It was illustrated above, however, that a hazard may have a combination of pathways. It is not possible to generalize about how to do exposure analysis because of the possible pathways and receptors of interest although there are a number of general models that can be calibrated for specific substances and situations.

Several types of atmospheric pollutants and their transformation are given in Geraghty and Ricci (1985). This source also gives a brief summary of some of the most commonly used models for dispersion of pollutants (particulates and gases) in the atmosphere and the application of the models. They give references for each model. Hanna et al. (1982) give technical details of models that are particularly applicable to effluents from energy systems. An earlier standard work is Perkins (1974).

The term "chemical fate" is often used to indicate the process of transformation of chemicals in the environment. A brief technical discussion of the mechanisms of chemical fate is given in Onishi (1985a). A review of the models, including the governing differential equations is given in Onishi (1985b). Among the models that are considered, one could mention radionuclide transport models, water quality models, groundwater models and mass transport models.

So far the description of exposure has been confined to toxic substances but other types of hazardous media also need to be considered. A propane storage tank, for instance, provides another type of hazard. Released propane first forms a cloud of vapourized propane which becomes mixed with the air. The cloud can become ignited creating a high temperature flame front and an explosive force. The wind and terrain will determine the location and shape of the vapour cloud.

A common release that affects both humans and the natural environment is the release of energy and material in the form of floods. In this case the pathway can be determined fairly well but the intensity and the presence of receptors is uncertain. A case occurred in Canada in which a herd of caribou were caught unexpectedly downstream of a hydro dam in high water when water was released with the start of the plant. The release was deterministic and the flood intensity could be treated deterministically for practical purposes. The uncertainty in this case was the presence of the receptors.

DATA SOURCES

So far we have concentrated on the models available to estimate the probability distribution of exposure intensity. Each of the models requires input data of various kinds. In other cases the exposure intensity can be estimated directly using historical intensities without an intermediate prediction model. Furthermore, the receptors need to be estimated so the exposure distribution can be expressed giving the number of receptors and the intensity of exposure. All of these estimates require data which are often difficult to obtain. Data are often collected nationally and are only applicable to the country of origin. Meteorological data is an example. In other cases, data collected in one country may, with care, be applied in another.

The models dealing with exposure through the air use meteorological data. In Canada that is available through the Atmospheric Environment Service of Environment Canada. They collect data, usually hourly, on some fifty meteorological variables and this is available in a data base. Wind speed and direction can be presented usefully as a wind rose. These are simply polar graphs that show the wind direction with arrows or bars of a length proportional to the frequency. Thus the windrose represents the probability of wind direction. The air transport models also use topographical data which is available through the Geological Survey of Canada. Accident data applicable to hazardous material transport are available from provincial and federal transport ministries.

CONSEQUENCE ANALYSIS

Consequences are the end results of human activities and natural events that we wish to avoid. In this section we discuss the consequences that result from exposure and describe some methods used to study the consequences from exposure. It will be seen that there is modelling uncertainty as well as the uncertainty inherent in the physical processes. These two types of uncertainty will be emphasized in this section.

There are three types of consequences: for the natural environment, for valued material objects, and for human health. This appears to be a clear and convenient classification but things are not that simple. Consequences have been treated as the end result of the chain of events. However, there may be some difficulty determining what should be treated as the "end point". An environmentalist might consider contamination of fish (through pesticide use on land) as the end result or consequence. A water quality engineer might stop earlier in the pathway and consider contamination of the water as a consequence. Medical people might go the other way and treat the health effect of those who eat fish as the consequence. In general, the logical stopping point in the pathway will be determined by the individual's interest, responsibility or discipline. The problem arises when we try to evaluate the risk because different people will want to treat different consequences as important. That issue, however, need not concern us in this section but should be understood if the product of technological risk analysis is to be of practical use in risk management.

HUMAN HEALTH CONSEQUENCES

Human health can be affected adversily if nutrition is insufficient or if energy to keep warm is withdrawn. Regulations or guidelines exist for both, e.g. the ambient temperature of buildings, or minimal nutritional requirements. There are also regulations or guidelines for the maximum amounts of toxic or carcinogenic substances in food, air and water. In this section we will describe one technique used to estimate risks from carcinogenic, toxic and mutagenic substances.

DOSE-RESPONSE MODELS

The response is the effect when a recipient is exposed to a substance under specified conditions and amounts in a single dose or over a given period of time. When a person is subjected to exposure to a toxic substance a full spectrum of consequences is possible but generally two levels are used in analysis.

> Acute toxicity - the amount needed in a single dose or exposure to kill the subject.
>
> Chronic toxicity - the effects arising from many repeated doses or repeated exposure over a long period of time. May be carcinogenic or non-carcinogenic.

Toxicologists use two forms of exposure. In one, it is supposed that a threshold exists. A receptor exposed to amounts below the threshold will not be affected. The dose is referred to as the "No Observeable Effect Level" often abbreviated to NOEL. In the second it is assumed that there is no threshold and the response probability, p, of the effect is estimated for the exposure. Generally, the dose for carcinogens uses cummulative exposure rates. Pollutants (e.g. lead) are often analysed using the threshold approach. It is not agreed among the experts which approach should be used for each substance.

Dose response models are a widely used means of estimating and representing consequences from exposure to toxic substances. A dose can be a single exposure or a daily intake of a given amount over a lifetime. A range of doses need to be considered. The effect is a consequence such as death or a particular condition experienced by a receptor. The receptor population may be restricted. As an example, if the potential consequence applies to unborn children, then the population might consist of pregnant women. Some members of the population will be susceptible to the substance and some will be resistant so, for a given dose, all will not be affected equally. A member of the population chosen at random may be one of the resistant or susceptible receptors so there is uncertainty about the effect on that member of the population. All of these ideas are combined in the dose-response model which given the probability of a randomly chosen member of the population suffering the consequence, the response, as a function of the dose. An example of a dose-response model is shown in Fig. 4. The dose is shown as d, and the effect by R. The response is the probability, p, of R as a function of the dose d, or, $P(R;d) = p$.

The dose-response model is useful for risk management. Suppose it is considered tolerable if one in one million of the population suffers cancer due to a toxic substance in the drinking water. This level establishes the probability at 1×10^{-6}, a point on the vertical axis. The corresponding dose is the point on the horizontal axis obtained from the curve. The dose is then set as the upper limit for the substance in the drinking water. It should be noted that a risk of one in one million may be the tolerable risk level resulting from all substances in the water. In this case the limits for each substance must be set so that the total risk is not exceeded.

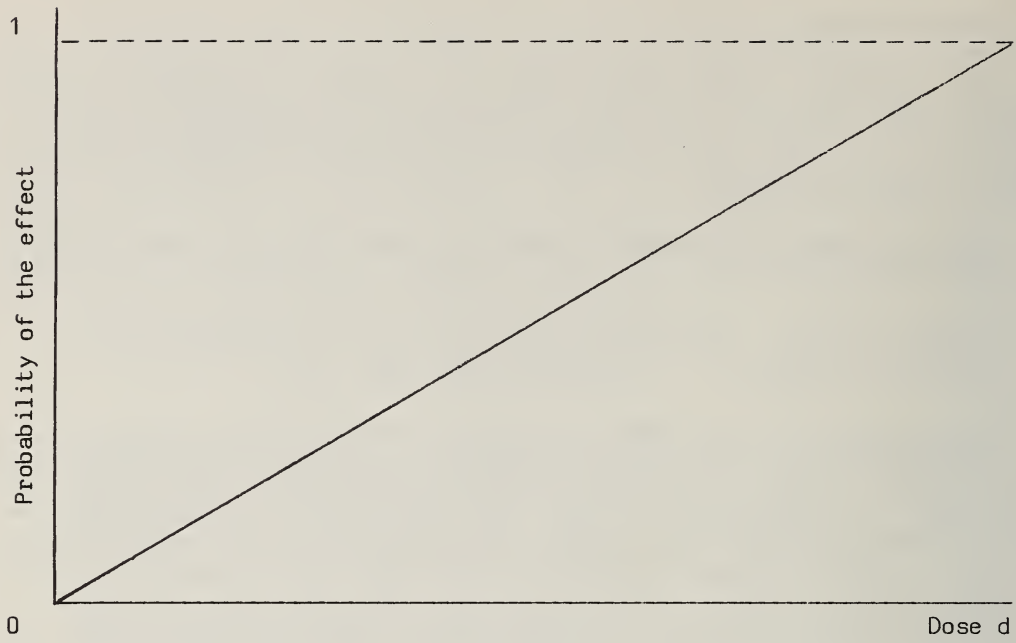

Fig. 4. Hypothetical dose-response curve

How is the dose-response curve estimated?. There are two broad approaches, epidemiology and toxicology. The epidemiologists approach the problem by gathering data on actual instances of exposure and response. Since we do not allow potentially fatal experiments on human subjects, their data are limited to what can be obtained by direct observation of exposure and inferred causes of death and morbidity. Occasionally there is an accidental release of material so the response to high doses can be observed but the actual dose received is usually not known precisely.

In studies of normal exposure, for example, air in the work environment, the dosage can be measured fairly accurately but the recipients cannot be controlled for other factors (smoking, genetic factors, dietary habits, previous work history). So even if a cohort group is used, it is difficult to isolate the effect of the particular exposure. For a more complete introduction see Kahn (1983).

The clinical approach is to use controlled experiments on animals. The animals are exposed to dosages measured, for example, in milligrams per kilogram of body weight. The exposure levels are much higher than would normally be experienced by humans so that the results can be obtained in the lifetime of the animals. This produces dose-response results for a limited range of doses. The further problem is extrapolating the laboratory results to much lower dose levels and from animal responses to human responses. Data will often fit several mathematical curves well in the range of observed measurement but diverge when extrapolated. A standard treatment of toxicology is given in Klaassen et al. (1986). The risk from toxic chemicals is treated in detail in Conway (1982).

CONCLUSION

This chapter has given an indication of what is involved in the technical treatment of risk. The general approach of this chapter shows how risk analysis contributes to EIA. Some simple examples have been used to illustrate the nature of the techniques and some references to further sources given. It will be evident that, to do a detailed risk estimation, requires the coordinated efforts of a number of disciplines and considerable resources. The release analysis stage may require reliability engineers for projects such as process plants. In contrast if the hazard is from an earthquake then volcanologists or geologists may be needed. The exposure analysis stage may require a combination of disciplines to develop the pathways and to estimate probabilities. Atmospheric scientists may be needed to trace the fate of chemical releases in air but may have to cooperate with hydrologists to follow the pathway into the water systems. Once the hazard has been received by the receptors then it is the turn of toxicologists or epidemiologists or others from the life sciences to estimate the effect. Clearly a detailed risk estimation for a significant project can not be done by a single individual but must be a team effort. In such cases it is necessary to manage the risk estimation activity so that the individual efforts are coordinated within the bounds of the study.

It will be evident also that there are many uncertainties in the estimation procedures. These provide the basis for challenge and debate by opponents to a project or activity. First, there is the inherent uncertainty in the process such as the wind direction when a release takes place or the susceptibility of a randomly chosen receptor to a given toxic dose. Then there is scientific uncertainty because the effect of a given cause may not be known. There are many new chemical substances that enter the market each year but there is reasonable scientific evidence about the effect of only a few of them. Finally, there is the uncertainty in the estimates such as the estimated probability of failure of a valve in a chemical plant. The result is that, even with the best intentions and management of the analysis, these uncertainties exist and can be a problem when the "objective" analysis is presented. This creates a quandary because professional integrity demands that the uncertainties should be recognized and yet by doing so the public confidence in the estimates will be shaken.

REFERENCES

Clark University Hazard Assessment Group. 1983. Methods for analysing and comparing technological hazards: definitions and factor structures. Report 3. Center for Technology, Environment and Development, Clark Univ., Worchester, Mass.

Conway, R.A. 1982. Environmental risk analysis for chemicals. Van Norstrand-Reinhold, New York.

Couch, W.J. (Ed.) 1983. Environmental assessment in Canada. Federal Environmental Assessment Review Office.

Geraghty, D. and P.F. Ricci. 1985. Mathematical models for atmospheric pollutants. In: Principles of Health Risk Assessment, (Ed.) P.F. Ricci, pp. 67-115. Prentice-Hall, Englewood Cliffs.

Hanna, S.R.,G.A. Briggs and R.P. Hosker. 1982. Handbook of atmospheric diffusion. U.S. Department of Energy, Washington, D.C. Technical Report DOE/TIC-11223.

Henley, E.J. and U. Kumamoto. 1981. Reliability engineering and risk assessment. Prentice-Hall, Englewood Cliff.

Kahn, H.A. 1983. An introduction to epidemiologic methods. Oxford Univ. Press.

Kates, R.W., C. Hohenemser and J.V. Kasperson (Eds.) 1985. Perilous progress: managing the hazards of technology. Westview Press, Boulder, Colorado.

Klaassen, C.D., M.O. Amdar and J. Doull (Eds.) 1986. Toxicology 3rd Edition.MacMillan, New York.

McCormick, N.J. 1981. Reliability and risk analysis: methods and nuclear power appplications. Academic Press, New York.

Onishi, Y. 1985a. Chemical transport and fate in risk assessment. In: Principles of Health Risk Assessment, (Ed.) P.F. Ricci, Chapter 4. Prentice-Hall, Englewood Cliffs.

Onishi, Y. 1985b. Chemical transport fate models. In: Principles of Health Risk Assessment, (Ed.) P.F. Ricci, Chapter 5. Prentice-Hall, Englewood Cliffs.

Perkins, H. 1974. Air Pollution. McGraw-Hill, New York.

Ricci, P.F. (Ed.) 1985. Principles of health risk assessment. Prentice-Hall, Englewood Cliffs.

Soloman, K.A., M.W. Meyer, J. Szabo and P. Nelson. 1982. Classification of risk. Report by UCLA School of Engineering and Applied Science, UCLA-ENG-8245.

U.S. Nuclear Regulatory Commission. 1975. Reactor safety study. Wash-1400. Washington, D.C., NUREG-75/014.

Veseley, W.E., F.F. Goldberg, N.H. Roberts and D.F. Hoasl. 1981. Fault tree handbook. U.S. Nuclear Regulatory Commission, Washington, NUREG-0492.

CHAPTER 5
METHODS OF TREATING BENEFITS IN RISK EVALUATION

Ronald Pushchak
Ryerson Polytechnical Institute, Toronto

Risks are not usually accepted voluntarily without a corresponding benefit. Whether the benefit is the thrill of scaling a mountain for the climbing enthusiast, the physical or psychological satisfaction produced by tobacco smoke entering the lungs of a cigarette smoker, or the additional dollars earned by workers in a dangerous occupation, some benefit is normally derived from taking each risk voluntarily. Since the level of perceived benefit is a significant factor in determining whether a risk is acceptable to the risk-taker, some estimate of the benefits associated with a risk, either explicitly or implicitly, must be made for each risk decision. The risk of climbing a mountain may be great, but if the perceived benefit is estimated to be greater, the risk will likely be taken. In other words, risk has little meaning independent of its benefit. Individuals voluntarily trade safety for some other benefit in their daily lives, which suggests that one of the central questions in risk evaluation is "How great a benefit do individuals require to accept voluntarily an additional risk to their health or life?".

THE TECHNOLOGICAL TREATMENT OF BENEFITS IN RISK EVALUATION

Although it is clear that the decision to take any risk is made in relation to the benefits it generates, in most risk studies the focus has been on risk analysis (the "cost" of risk) rather than benefit estimation. Increasing efforts have been made to calculate the probabilities of an undesirable event occurring or to estimate the magnitude of its consequences, all the while assuming that the benefits of a project, either in kind or in distribution, exceed the risk. Generally, benefits have been less rigorously estimated than risks. Risk studies have typically assumed that if objectively measured risks can be demonstrated to be low in absolute terms, benefits need not be estimated in detail nor their distribution established.

However, recent attempts to site hazardous waste facilities or to approve pesticide use through risk analysis (Kunreuther and Linnerooth 1983; Johnson 1982) suggest that this assumption is no longer tenable. This is due in part to the lack of replicability and credibility of risk analysis results (Kunreuther and Lathrop 1981; Blokker 1981). This uncertainty has raised questions as to whether decisions can be made independent of benefit considerations. Increased attention to the benefit side is also due to the high levels of public opposition to risk generating projects. The Ontario Waste Management Corporation's (OWMC) first site an industrial waste facility in southern Ontario has met with considerable opposition in the past (Farkas 1982) and the OWMC is experiencing growing opposition from the present candidate site. Similarly, opposition to the Darlington Nuclear power station and to the proposed movement or storage of soils contaminated with low-level radiation in Scarborough have been noted. High levels of public opposition and the resulting lengthy siting process have drawn attention to the lack of a clear relationship between the risks generated and the benefits associated with such proposals (Kunreuther and Linnerooth, 1984). Consequently it has become increasingly necessary that the benefits of a project be explicitly estimated and that the decision process be broadened to achieve a more equitable sharing of potential benefits and risks.

This is particularly necessary if Risk Assessment (RA) principles are to be specifically included in Environmental Impact Assessment. Most EIA procedures require that the benefits of a project be estimated (as part of the need for the project) in addition to the identification and measurement of impacts. If RA is to be a part of EIA, the benefits of a project and their distribution will have to be explicitly stated, particularly in cases where low probability-high consequence risks are to be balanced against benefits.

THE NATURE OF BENEFITS

It is important to distinguish between the benefits of risk-taking (effects that compensate people for taking risks) from the benefits of reducing the level of risk. The focus of this paper is the measurement of the benefits of risk taking. Benefits in an EIA or RA study are the desirable results of a project, the ultimate gains for which any action is taken. If these benefits could be accurately measured, the Risk Assessment process would be a simple matter. However, the methods of estimating benefits created by a risk-related project are limited in accuracy because of three central analytical problems:

1. identifying the appropriate benefits;
2. benefit measurement; and
3. determining the expected distribution of benefits.

The same difficulties apply to risk identification and measurement (the cost side of risk). However, more efforts have been made to assess risk in itself rather than the associated benefits.

BENEFIT IDENTIFICATION

Commonly, the benefits of a project are identified as being either direct or indirect. Direct benefits are considered to be the primary outputs of a project (the electrical power produced by a coal-fired generating station or the ability to contain industrial wastes securely in a waste disposal facility). Direct benefits ideally measure the achievement of the goals of the project (to produce power or contain wastes). There are, however, indirect benefits generated by each project which are more difficult to identify. Indirect benefits are those desirable effects created by the project either by stimulating the production of related goods and services or by the increased demand for goods resulting from project expenditures.

If a liquid natural gas storage facility is taken as an example, the project, while creating the risk of an LNG explosion, produces a direct benefit in the storage of imported natural gas for domestic supply. It creates indirect benefits by stimulating the growth of gas-related industries near the plant (petrochemicals) and by expenditures on local goods and services in constructing and operating the plant.

The difficulty in estimating indirect benefits is one of limiting their number because the set of indirect benefits can be considerably expanded to include: the multiplier effects of increased employment in the community, the growth effects of reduced local tax rates as a result of LNG plant tax payments, or the subsequent reduction in air pollution damage resulting from the conversion of coal to gas fired power plants. The same holds true for indirect benefits related to risk. It might be argued, for example, that the LNG plant would reduce the risk of occupational deaths in the coal mining

industry by reducing the demand for coal, or that the risk of lives lost due to air pollution would be diminished through conversion to natural gas producing a net improvement in social welfare.

While all such indirect benefits are the result of the project, their link to actual environmental impacts or risks becomes less plausible the more indirect their relation to primary benefits becomes. Critics of the EIA process (McAllister 1980) have pointed out that an expansion in the number of indirect benefits can substantially affect the EIA or risk evaluation outcome, particularly in cases where primary benefits alone may not have been sufficient to offset the impacts. Consequently, if risk analysis is to be integrated within an EIA framework, the scope of indirect benefits would have to be limited in a manner similar to current EIA scoping methods.

BENEFIT MEASUREMENT

For purposes of comparison, the benefits and risks of a project are measured in relation to a common scale and, in most cases, it is the value of the benefits in dollars. Fortunately, the direct benefits of a risk generating project are usually goods exchanged in the marketplace (kilowatts of electrical power, cubic feet of natural gas) and therefore can be valued empirically using market prices for those goods. However, difficulties in measuring benefits arise because many direct and indirect benefits are intangible, extra-market goods for which no true dollar value can be derived empirically (a recreation experience or increased life expectancy). This does not mean that intangible benefits cannot be valued subjectively, only that their value must be imputed from sources other than the market.

The evaluation of the cost associated with enduring risk is also constrained by this measurement problem; for example, the benefit of a risk management study may be a reduced level of human exposure or the number of potential lives saved. The value of a human life cannot be empirically determined because human life falls outside the conventional market process; consequently, alternative non-market methods of valuing the benefits of risk reduction have become necessary. However, there is little consistency in the literature concerning the methods used to estimate the benefit of reducing risk (Graham and Vaupel 1981). Several methods have been used ranging from values of life revealed by human behaviour to the value expressed by survey respondents. The most frequently used methods of benefit measurement are:

1. Revealed Life Value
2. Direct Life Value
3. Willingness to Pay for Risk Reduction
4. Willingness to Accept Payment for Risk

REVEALED LIFE VALUE

This measure of risk benefit is derived from the revealed preference to take risks first suggested by Starr (1969). In theory, individuals reveal the true value or benefit of taking a risk through their behaviour rather than through direct statements regarding risk. Starr (1969) argues that over time, society has arrived at a reasonably acceptable balance between the risks and benefits of an activity. Consequently, the value of a benefit can be measured by observing the relationship between risks presently taken and benefits received.

Following Starr's (1969) concept, the value of a life saved has been estimated using the wage differentials between risk and non-risk generating occupations. It is argued that by accepting a higher wage to take extra risks, individuals have implicitly revealed information about the value of their life (Thaler and Rosen 1976; Smith 1979; Viscusi 1979). In Table 1 the values of small risk increments have been extrapolated to compute the value of a life.

Table 1. Implicit values of life (thousands of dollars 1987)

	Source of Evidence	Author	Value of a life[1] 1987	Risk reduction
1.	Blue collar workers in manufacturing and construction	Dillingham (1980)	$523	10^{-4}
2.	Workers in risky occupations	Thaler and Rosen (1976)	$683	10^{-3}
3.	Males in manufacturing industries	Smith (1979)	$3,850	10^{-4}
4.	Blue collar workers	Viscusi (1979)	$3,898	10^{-4}

[1]all values converted to 1987 dollars using the Consumer Price Index
Source: Adapted from Blomquist 1982:35

The variation in values in Table 1 indicates that there are major discrepancies between studies estimating life value and subsequently that revealed life values are not reliable measures of risk benefits. Blomquist (1982) suggests that variations in revealed value are useful only in measuring marginal risk changes because the extrapolation from a small risk change to the full value of a life tends to significantly underestimate the value of life.

A central problem in revealed value measures is that they assume individuals make free choices about their occupation, have reasonably perfect knowledge about the availability of jobs elsewhere and experience no barriers to mobility. However, these assumptions are untenable, particularly in the last decade of high unemployment when labour supply far exceeded demand. Under such conditions, and given that families make fixed financial and social investments which reduce mobility, labourers often have little choice in occupation or the risk associated with working conditions.

DIRECT LIFE VALUE

The direct quantification of the value of a life is an attractive method of measuring benefits because it permits a cost-benefit comparison between risks and benefits. This approach does not use preferences as the basis of valuation and, therefore, does not represent an "economic" value that is comparable to other economic values which are typically based on preferences. Although direct measures of life value have been criticized because they

dehumanize life, something that is argued to be intuitively infinite in value, direct values have been commonly used to establish a balance between risks and benefits.

The conventional method of direct valuation has been the calculation of foregone earnings per life lost assuming that each life, if lived to its expected extent, would have produced wages that would have benefited society. Typically, estimates of foregone earnings have produced life values ranging from $200,000 to $350,000 (May 1982). Graham and Vaupel's review (1981) of 35 risk assessment studies found that 15 of 24 cost-benefit risk analyses used a foregone earnings measure which produced an average life value of $217,000.

Foregone earnings measures tend to produce values that are intuitively low in comparison to revealed life values because the estimation of future earnings does not take into account non-wage benefits produced in a lifetime or the happiness and psychological benefits to others of a life completed. In addition, the foregone earnings measure tends to discount the life value of elderly, unemployed, and poor individuals whose expected earnings at the time of measurement are low. Children have an uncertain foregone earnings value bacause their occupational choice has not been made and those not paid wages for their labours (volunteer workers) also have an uncertain value.

WILLINGNESS TO PAY FOR RISK REDUCTION

To overcome the deficiencies in foregone earnings estimates, a measurement which determines the value of risk reduction by surveying a sample of the population at risk has been put forward by Schelling (1968) and Mishan (1982). Theoretically, the value of a reduction in risk is reflected in the willingness of the recipient to pay for that reduction rather than accept a higher risk. It assumes individuals treat additional years of life as any other good and are willing to trade them for other benefits. Following this model, respondents have been surveyed for their willingness to pay for risk reductions in health services, nuclear power plant operation and other risk related enterprises by creating a hypothetical market for benefits.

Blomquist's (1982) review of willingness to pay studies for reductions in heart attack risk, airline accident risk, and nuclear power plant risk indicates that life values estimated were considerably higher than foregone earnings with values for risk reductions of 10^{-5} to 10^{-6} varying from $3,372,000 to $10,120,000 and for reductions of 10^{-3} from $57,000 to $62,000. These results are consistent with Graham and Vaupel (1981) who found in seven studies willingness to pay values three times greater than foregone earnings with a willingness to pay median life value of $625,000 dollars. Blomquist suggests that the results show an expected pattern of decreasing life values as the size of the risk reduction grows larger indicating the diminishing marginal value of an additional risk reduction. Initial reductions in risk are perceived as very valuable whereas additional reductions are less valuable. This suggests that implicitly, zero risk is low in value because some (low) level of risk is accepted as a part of life.

It has also been suggested (Hyman 1981; Blomquist 1982) that the differences between values for different types of risk, for example between airline and nuclear reactor accidents, reflect the perceptual aspects of each risk. Willingness to pay measures tend to vary with the dread nature of the risk, its common properties and whether it is voluntarily accepted by the respondent. In this light, the willingness to pay for risk reductions is not a

linear function. Rather, it can be expected to change over the range of risk magnitudes and risk types.

There are, however, limitations in the reliability of willingness to pay measures that stem from the survey techniques used. Surveys assume rational, truthful response and consistency among responses. Freeman (1979) suggests that the measures derived are not free from inconsistency or bias because there is little incentive for respondents to be accuarate in answering survey questions. Respondents may overstate their willingness to pay assuming that the response will effect a reduction in risk and that others will actually pay for risk reduction programs (strategic bias). Respondents may also overstate their willingness to pay because not doing so will suggest that they care little about reducing risks. Consequently, respondents may give responses they feel the surveyor would like to receive (instrument bias). Moreover, respondents may not take a hypothetical market seriously and may give responses that are unrelated to their true values (hypothetical bias).

One additional limitation of willingness to pay measures is that they are strongly influenced by the present distribution of wealth. Respondents may understate the benefits of a risk reduction because their responses are constrained by their disposable income. If the respondents are poor and if they were able to express their preferences independent of their income, the computed value of benefits might be greater.

WILLINGNESS TO ACCEPT PAYMENT FOR RISK

Measures that survey willingness to accept payments for benefit reductions rather than willingness to pay for additional benefits are a recent refinement of survey estimates (Meyer 1976; Kelly 1980). Respondents are asked the amount they would accept to experience a loss in amenities or an increase in risk. In this way, the constraint of income is removed and additional value not related to the respondent's economic well being can be expressed. Meyer, in his studies of recreation benefits for residents near the Fraser River, found that the willingness to accept payment amounts were much higher than willingness to pay measures for the same environmental impacts (Table 2). The large difference between the two measures indicates the difference accounted for by the income constraint whereas the wide confidence intervals suggest substantial variations in responses among individuals.

Table 2. Comparison of willingness to pay and willingness to accept payment measures

Question Type	Mean $	95% Confidence Level $
(a) Annual Amount you would pay	1,099	± 993
(b) Annual Amount you would accept	20,961	± 6,028

Source: Meyer 1976

For willingness to accept payment measures, similar problems of instrument bias and hypothetical bias exist. In addition, strategic error is more likely to occur resulting in overestimation of benefits because some individuals may demand large or infinite amounts for additional risk which would drive the mean benefit values upward. All benefit measures, however, are subject to one central constraint; the limited meaning of extrapolating benefit measures for small risk increments to the total value of a life. The relevant benefit measure in risk evaluation is not the value of life or death but the benefit of a small change in the risk of death associated with a specific project, which for most people is finite (Linnerooth 1982). Instead, Mishan (1982) suggests risk assessments should avoid models that seek to produce an average, all-purpose value of life and strive for methods that value an increment of risk for a specific project, place, and time.

Finally, the foregoing discussion of benefit measures raises an important question in risk evaluation; who values the benefit, the analyst or those affected by risk? Prior to the development of willingness to pay and willingness to accept payment measures, benefit estimates were made by those other than the population at risk. However, in the present climate of public opposition, benefits based on foregone earnings or wage differentials would not likely be accepted by residents living near a liquid industrial waste or liquid natural gas storage facility. The literature suggests that willingness to pay or accept payment measures are being increasingly used in risk evaluation because they take into account the non-economic (psychological and social) benefits of risk reduction and also because they are more consistent with the basic welfare economics model of risk evaluation which assumes that each individual is the best judge of his or her own welfare. These measures suggest that only a member of the population at risk can specify the utility of a reduction in risk, a reasonable assumption in keeping with the conventional wisdom that the recipient of a gift is the arbiter of its value.

BENEFIT DISTRIBUTION

All benefits and risks of a project are not distributed equally, either spatially or through the economic and social strata of society. In a spatial context, benefits can be divided into two groups; local and external benefits. Although the point of division between the two is subjective, local benefits are those realised by the immediate community whereas external benefits accrue to regional or national populations. For some risk activities (food additives, pesticide use) this distinction does not apply because the distribution of benefits is broad and roughly equal. However, with risk generating facilities at a specific site, it is usually the case that external benefits are the primary outputs of the facility (electric power produced or space for wastes) and are broadly distributed throughout the region whereas local benefits are primarily indirect benefits (the stimulation of production or demand for local goods) that are confined to the vicinity of the facility. The difficulty arises in specifying the group which considers a risk acceptable.

This is particularly difficult in the case of risk-generating facilities because often the risks and benefits are not equitably distributed in a region. While a facility may produce primary benefits that are broadly distributed across a region, often it fails to produce local benefits. The obvious example is an industrial waste storage facility which does not stimulate local production or employment and requires little in the way of locally produced goods or services. Risks, however, are indirect costs that are usually spatially limited within the local area. Consequently, it is common that the

local community is asked to accept spatially confined risks with few local benefits while others elsewhere in the region receive external benefits. Moreover, the external benefits, although large in total, are diffused throughout the population. Each actor in the region has only a small amount at stake in the risk/benefit decision but the per capita benefits and risks for members of the local community are larger because both are distributed among a small number of residents. Each member of the local community therefore has a large amount at stake in the risk/benefit decision (O'Hare 1977).

Conventional risk evaluations have neglected the distribution of benefits because aggregate external benefits typically exceed social costs, consequently net social welfare is considered to have been improved and all of society is believed to benefit. However, bacause a spatial unequal distribution of benefits and risks tends to occur more frequently for risk-generating facilities, the need to specify the distribution of local and external benefits in a risk evaluation increases.

EVALUATION OF BENEFITS

Benefits are evaluated in current risk assessment practice from two perspectives; benefits to the proponent and general societal benefits. In the ideal case, risk-related actions which produce benefits for the proponent also provide benefits for society at large. However, in some cases, proponent benefits can be inadequately balanced by societal benefits and the decision to accept a risk can result in an inequitable allocation of risks to society with private benefits being generated by public risk.

Benefits to the proponent

Benefits resulting from a risk-related decision which accrue to the proponent are direct benefits that meet the goals set out for the project. For a nuclear power station, the proponent's benefit is the profit derived from the sale of electric power produced and for a liquid natural gas storage facility, the benefit is the profit derived from the sale of stored natural gas. In most cases, the benefits to the proponent are measured in terms of commonly exchanged goods (e.g. amount of natural gas sold) that have a market price.

The evaluation of proponent benefits is usually an internal calculation with benefits measured by the net return on investment. The investment return is balanced against the costs of production, capital, labour and other costs (waste disposal, risk of liability) to determine whether benefits exceed costs. In this way, benefits to the proponent are usually considered private benefits because the proponent is able to choose the level of benefit to be accepted, independent of public scrutiny.

Benefits to society

Risk-related projects also tend to produce a set of direct and indirect societal benefits that are public rather than private in nature. Societal benefits include direct benefits such as an available supply of natural gas for domestic consumption or a reduction in the cost of electricity. However, the benefits most difficult to measure and evaluate are indirect, collective benefits such as increases in employment, regional economic growth effects, or an increased demand for local goods and services. Societal benefits have some

of the properties of public goods in that no individual can choose the level of benefit they will consume. Rather, for societal benefits such as an improvement in air quality or a reduction in risk, the same level of benefit is produced for all. Similarly, no one can be excluded from enjoying the benefit. Once a societal benefit is provided for one person, it is available to all.

Because individuals are not free to choose whether to accept a collective benefit or to determine what level of benefit to consume, the imposition of societal benefits and associated risks is involuntary once a risk-related choice is made. Similarly, because the benefits are collective rather than private, the decision to impose societal benefits or risks is necessarily a public decision. Consequently, two risk-benefit evaluations tend to be done for each project; the internal evaluation of the proponent with regard to private benefits and a public evaluation of collective risks and benefits. Where the two evaluations produce contradictory results, a maldistribution of risks and benefits can occur depending on which evaluation prevails.

May (1982) describes the conflict between a risk-evaluation done by the Ford Motor Company and public evaluation of risk for the Pinto automobile. In the early 1970s, Ford engineers discovered the fuel tank of a Pinto tended to burst and ignite if hit from the rear with a subsequent potential loss of life. They were faced with a decision of redesigning the fuel tank or leaving it unchanged and resorted to a cost-benefit analysis to determine the costs of redesign and the risk reduction benefit to society.

Ford estimated its unit cost of redesign was $11 dollars per automobile, which for 11 million cars and 1.5 million trucks yielded a total redesign cost of 137 million dollars. Their estimate of the benefits of avoiding risks through redesign was based on a foregone earnings value for a human life of $200,000 per death, plus $67,000 per injury and $700 material damage to the vehicle. The resulting benefit for 180 estimated deaths, 180 serious burn injuries and 2,000 burned vehicles was 49.5 million dollars (May 1982:37). Ford decided, on the basis of this risk-benefit evaluation, to leave the vehicle unchanged and allow the benefits (a slightly cheaper vehicle) and risks (potential deaths and burn injuries) to be borne involuntarily by society.

May (1982) points out that the foregone earnings measure of a life in the Pinto analysis substantially undervalued its worth in common societal terms and argued that if Ford has used the high estimate death value computed by National Highway Traffic Safety Association (NHTSA), the benefit total would have been $161.2 million dollars, a figure considerably higher than the cost of redesign. However, beyond the imprecision of the benefit estimate, the Ford case indicates that a benefit-risk decision made by the proponent did not include a public decision involving public risks. This resulted in the involuntary imposition of uncompensated risks on consumers.

NEGOTIATED BENEFITS IN RISK EVALUATION

The difficulties in estimating the social benefits of a project and the associated problem of the unequal distribution of benefits and risks have resulted in a series of failures in the management of risk. Projects have been rejected or delayed and the level of opposition has been extreme suggesting that the risks involved were unacceptable. While it is clear that individuals voluntarily and routinely trade some reduction in safety for monetary or other benefits, the indirect or hypothetical measures of life value commonly used in risk assessments have not been helpful in deciding which risks are acceptable,

to some extent because the benefits estimated are abstract but also because benefits have commonly been estimated for society in general rather than for those bearing the risks.

The question which bears directly on the acceptability of a risk is: how much do affected inividuals require as an <u>a priori</u> benefit to voluntarily accept a small additional risk of death? At low levels of risk, small increases in risk would intuitively require smaller levels of compensation, whereas at greater risk levels, an equal increase in risk would likely require greater benefits. Schulze and Kneese (1981) have expressed this risk benefit balance as a non-linear relationship. For small increases in risk (.001) and at low risk levels, a small level of benefit would be required. For example, the required benefit in wages for risk of .001 varies between 340 and 1,000 dollars per year (Smith 1973). However, as risk increases toward unity (the certainty of death), the demand for benefits would increase dramatically tending toward infinity to avoid that certainty. In this way the incremental benefit demanded would increase with the seriousness of the risk.

Conventional measures of benefit have focused on the right side of the relationship in estimating the benefit value at loss of life. However, Fig. 1 suggests that intuitively, the value of life is infinite at that extreme and that values less than an infinite amount would be unacceptable. Moreover, for typical risk assessments, the issue is never the certainty of loss of life but a change in risk levels at the left side of the X-axis in Fig. 1. Consequently, the benefit measures relevant to the decision are not life or death but the value of a small change in mortality risk, which for most individuals is finite. Howard (1981), for example, suggests that, for a considerable range of risk (up to a 1 in 1000 chance of dying per year), the perceived cost of each risk increment is finite and increases gradually, and that in this range, individuals will accept risks if benefits are commensurate.

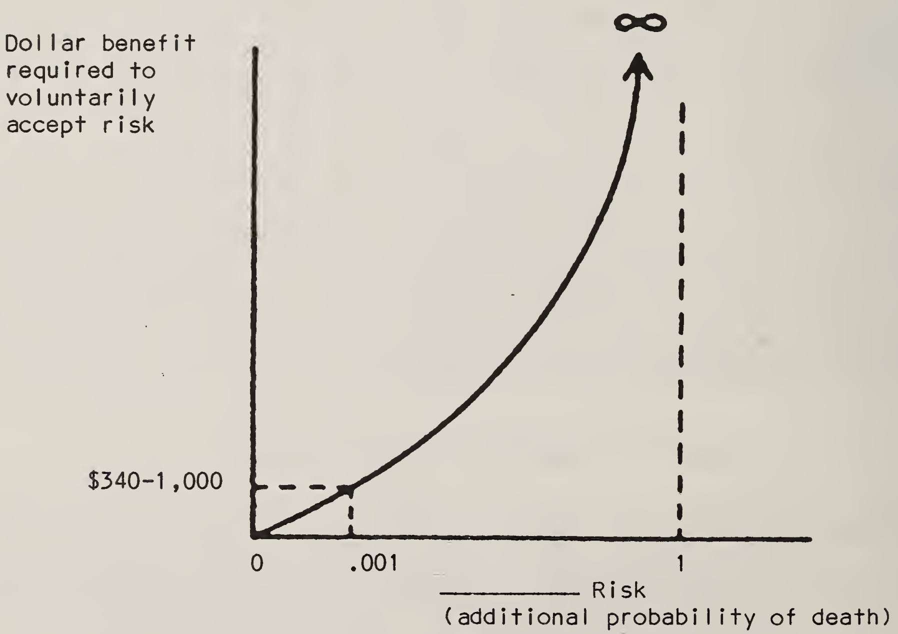

Fig. 1. Risk-benefit relationship. Source: Schulze and Kneese 1981

Recent risk studies (Mishan 1982; Kunreuther and Linnerooth, 1984; Sorensen et al. 1984) have suggested that the required benefit be measured not hypothetically, but by direct negotiation with the affected community to determine an acceptable level of benefit. This method of benefit measurement is of increasing interest in risk assessment studies because it directly addresses the unequal distribution of benefits that occurs in conventional risk assessment studies. While societal benefits for a project may, in aggregate, be large, many risks will be borne by a subset of the population or a smaller affected community including risks to health and safety, or changes in economic and social well-being.

While conventional estimates of benefits and losses are conducted by risk assessors (scientists and engineers), the direct negotiation of benefits assumes that members of the affected community are able to decide what benefits are in their best interest. Green (1975) has argued that although risk analysts are competent to identify and quantify potential risks and to identify potential benefits:

> No elite group of experts, no matter how broadly constituted, has the ability to make an objective and valid determination with respect to what benefits people want and what risks people are willing to assume in order to have those benefits (Harold P. Green, cited in Rowe 1981:89).

By compensating the affected portion of the community, at a level determined through direct negotiation, it is possible to redistribute the benefits from the broader society to the affected population and reward people for taking risks from which others benefit. This is what Kunreuther and Linnerooth (1984) call "broadening the agenda" for each stakeholder by more explicitly sharing the potential losses and gains of a project.

However, where benefits and losses are negotiated directly, there are several conditions that must be met to assure an accurate estimate.

Information A community requires access to sufficient information about the levels of risk (a risk analysis), the costs and the benefits generated by the proposed facility. Compensation negotiated without sufficient information about potential risks might be too low, whereas benefits determined without knowledge of the expected costs of the project might be overvalued.

Voluntary Acceptance Communities require free choice in the determination of benefit levels and in accepting benefits. If benefits are imposed from outside the community or if the community is coerced to accept them, the settlement negotiated will not likely be an accurate measure of benefits.

Morally Relevant Community Negotiations should be conducted with the morally relevant community, that group of individuals or their representatives which will be affected by project risks. Negotiations with unaffected communities can result in benefits being received by those not bearing risks, thus maintaining the unequal distribution of risks and benefits.

In the Maple, Ontario solid waste disposal case (Cameron and Bordessa 1982), in which a municipal waste facility was proposed for an exhausted gravel pit, it became clear after protracted hearings that risks and benefits could not be easily quantified. Consequently, after levels of risk had been technically estimated, the determination of project benefits to the local

community was left to negotiation between the proponent and regional representatives. Benefits negotiated by the political representatives included:

(a) a "tipping" fee of 30 cents per ton of waste with initial receipts used to construct a by-pass road around the Town of Maple;
(b) free dumping of waste for the residents of the town up to a population of 50,000;
(c) conveyance of the completed landfill and its buffer areas to the Township free of charge after the site was filled.

This monetary compensation was, in addition to other non-monetary compensation, imposed by the Ontario Ministry of the Environment in its Approval Certificate which included a monitoring program, connections to the regional sewer system, and the provision of a performance bond of $100,000 to deal with claims arising from off-site impact.

Similar benefits have been negotiated in both North America and Europe for a variety of risk-generating facilities. Bjornstad and Goss (1981) noted that payments in lieu of taxes have been made to communities in the United States when siting high-level nuclear waste repositories. The benefits had the effect of lowering existing taxes or reducing the growth in tax rates for communities accepting government operated facilities. Similarly, direct payments have been made to host communities in the U.S. and reductions in electricity rates of 15-20% have been obtained by residents living within 15 km of a nuclear power plant in France (Kunreuther and Linnerooth 1984).

Two other cases cited by Kunreuther and Linnerooth (1984) are particularly relevant examples of negotiated benefits which offset potential risks. In Wyoming, the construction of a 1500 megawatt coal-fired power plant had been halted by an injunction because of the potential risk of environmental damage. The controversy was resolved when the utility companies agreed through negotiations with environmental representatives to establish a 7.5 million dollar trust fund to be used for the purchase and preservation of a 60 mile segment of the Platt River which was habitat for migratory fowl including the whooping crane. After the settlement had been negotiated, construction of the power plant was allowed to be completed.

In the Federal Republic of Germany, the STEAG utility company in 1976 proposed a 1400 megawatt coal-fired power plant for the City of Bergkamen. A local citizens' group opposed the project and threatened to delay the licensing process. The utility company negotiated a settlement with representatives of the citizens' group which included a DM1.5 million payment in return for a voluntary end to citizen opposition. The validity of the contract was tested in federal court when the City of Bergkamen refused to pass on the payment to the citizens' group. However, the court decided that the negotiated agreement was valid and that the citizens' group was entitled to be compensated for its perceived risks.

THE NATURE OF NEGOTIATED BENEFITS

An important aspect of negotiated benefits is that in many cases, they need not be monetary. Although, monetary compensation is usually a part of negotiated agreements, the Maple, Ontario and Wyoming cases suggest that the benefit to those affected by a facility can be expressed in non-monetary means; monitoring, physical design changes, or local citizen participation in

operating decisions. Studies by Kelly (1980) and Carnes et al. (1982) of risk acceptance behaviour indicate that in cases where the primary benefit is perceived to be a reduction in risk to the affected population, non-monetary compensation tends to be a major part of the negotiated outcome.

A second aspect of negotiated benefits is that in cases where a risk continues to affect a specific population over time (e.g. those living near a hazardous waste disposal facility or an LNG plant), benefits have been negotiated which are designed to run with the risk over time. In this way, a "stream of benefits" is initiated which continues to compensate individuals for a continuing risk. Monetary benefits negotiated and paid at one time are less able to compensate an exposed population for a continuing risk than monetary and non-monetary benefits which generate a stream of benefits over time. Arrangements such as payments in lieu of taxes, reductions in utility costs, tipping fees, access to landfill sites, and continuous monitoring programs are examples of methods negotiated to produce a benefit "stream". Moreover, unlike conventional benefit measures, benefits determined through negotiation need not be estimated beforehand as part of risk analysis. This relieves the risk analyst of the difficult burden of measuring benefits (discussed above) and the implicit valuing of human life that is required in a conventional risk assessment study. Instead, the negotiations attempt to determine how much individuals will accept to bear the additional risk indicated in the risk analysis process.

Negotiated benefits are increasingly attempted in decisions involving risk because the transfer of benefits satisfies the basic utility maximizing rationale underlying public choices about risk. If the proponent makes benefit payments, financed from the project and equal to the social or risk costs of a facility to those who are affected, and still produces a surplus, the social welfare of society is assumed to be improved. The determination of benefits alone does not unambiguously increase social welfare unless the compensation is actually paid, so everyone is really better off. If the benefits have been freely negotiated and accepted by the community, it can be assumed the compensation is adequate, that is, it reflects the true risk benefit for the community. Conventional measures of benefits have been found to vary considerably between populations and across different risk types, consequently, the reliability of those benefit measures has been questioned, particularly for estimates of human life value. Negotiated benefits, however, assume that the outcome is determined by those affected and with sufficient information about the nature of the risk so that the benefit level is likely to balance the perceived costs to the community of the risk increment.

Some communities have recently begun to express an interest in negotiating to obtain typically unwanted facilities because of the potential benefits that might be obtained through negotiations. Elliot Lake, Ontario, expressed its desire to negotiate the location of a low-level radioactive waste disposal site in January 1988 because of the "potential economic benefits" of the facility including the possibility of a research centre for monitoring and studying wastes and the associated prospect of increased employment and economic diversification. A set of communities, including Chalk River near Ottawa made a similar proposal a few weeks later suggesting that benefits freely negotiated and accepted by a community can offset perceived risks. While it is true that both communities have had extensive experience with nuclear power and waste and depend on the nuclear industry for their economic well-being, it is also clear that the estimation of benefits by each community is the major factor influencing the community's willingness to accept additional risk.

In addition, a negotiated benefit implies that those bearing a risk have consented to do so and that the risk is no longer borne involuntarily but by choice. Derr et al. (1983) argue in their study of compensation for risks in the workplace that increasing the degree of consent improves the acceptability of a risk through realistic compensation. Their recommendation, based on calculations for the nuclear industry, was that high-risk workers should receive special hazard compensation of $1,000 per person-rem paid directly as wages, in addition to special education and training in radiation health risks (non-monetary benefits). They argued that such a compensation package could produce the free and informed consent necessary for the voluntary acceptance of risks.

CONCLUSIONS

If the environmental impact assessment process is to include a risk analysis component, it is clear that a method of estimating benefits associated with risks will have to be incorporated. The foregoing review of benefit estimation methods suggests that the direct measures of benefit, revealed life value and direct life value, have produced benefit estimates that were inconsistent among populations or risk types and tended to underestimate life values in risk decisions. The inconsistencies among benefit estimates have called the reliability of direct methods into question and the under-estimation of life values has tended to reduce the public acceptability of risk-generating facilities by suggesting benefits that are intuitively too small in relation to perceived risks.

Hypothetical willingness to pay and willingness to accept payment methods are considered to produce more realistic measures of benefits because they implicitly include some of the intangible family and social values of life not measured in direct methods. Of the two, the willingness to accept payment method is considered to most accurately reflect the perceived value of a benefit because it imposes no income constraint on a person's willingness to accept payment for bearing a risk. The literature on risk benefit analysis suggests that the willingness to accept payment measure is the most credible method of estimating benefits and would be applicable to an EIA in judging the acceptability of a risk.

Finally, the literature on benefit estimation reveals that where benefits can be negotiated with an affected population, they need not be estimated as part of either a risk analysis or an EIA. Negotiated benefits have been useful in improving the acceptability of a risk by redistributing benefits from the source of risk to those exposed to it and by allowing for the negotiation of both monetary and non-monetary benefits. A process of negotiating benefits for risk impacts could be used as part of an EIA process similar to the present practice of environmental mediation for social and economic impacts.

ACKNOWLEDGEMENTS

I would like to thank the two referees for their constructive comments and suggestions which added substantially to this paper.

REFERENCES

Bjornstad, D. and E. Goss. 1981. Measuring the impacts of using payments in lieu of taxes to compensate communities when siting high-level, nuclear waste isolation facilities. Oak Ridge National Laboratory, Tennessee (draft copy).

Blokker, E.F. 1981. The use of risk analysis in The Netherlands. Angewandte System analijse Band 2, pp. 168-171.

Blomquist, G. 1982. Estimating the value of life and safety: recent developments. In: The Value of Life and Safety, M. Jones-Lee (Ed.). Geneva, North Holland, pp. 27-40.

Cameron, J. and R. Bordessa. 1982. Solid waste disposal: risk assessment or risk management? The pits at Maple, Ontario. In: Living with Risk: Environmental Risk Management in Canada, I. Burton, C.D. Fowle and R.S. McCullough (Eds.). Institute for Environmental Studies, University of Toronto, EM-3:129-143.

Carnes, S.A., E. Peelle, E. Copenhaver, J. Reed, E. Soderstrom, J. Sorensen and D. Bjornstad. 1982. Incentives and the siting of radioactive waste facilities. Oak Ridge National Laboratory, Tennessee.

Darby, Wm.P. 1979/1980. An example of decision-making on environmental carcinogens - the Delaney clause. J. Environ. Systems, 9(2):109-121.

Derr, P., R. Globe, R, Kasperson and R. Kates. 1983. Responding to the double standard of worker/public protection. Environment, 25(6):6-36.

Dillingham, A.E. 1980. The relationship between estimates of wage premiums for injury risk and the measurement of injury risk: results from one population. Department of Economics, Illinois State University, September 1980.

Farkas, E. 1982. The NIMBY syndrome. Alternatives, 10(2 and 3):47-50.

Freeman, A.M. 1979. The benefits of environmental improvement. Johns Hopkins Univ. Press, Baltimore.

Graham, J.D. and J.W. Vaupel. 1981. The value of a life: what difference does it make? In: Risk-Benfit Analysis in Water Resources Planning and Management, Y.Y. Haimes (Ed.), Plenum , New York, pp. 233-243

Green, H.P. 1975. The risk benefit calculating in safety determination. George Washington Law Review, 43(3):791-804.

Howard, R.A. 1981. The risks of benefit-cost analysis. In: Risk-Benefit Analysis in Water Resources Planning and Management, Y.Y. Haimes (Ed.), Plenum, New York, pp. 135-148.

Hyman, E.L. 1981. The valuation of extra market benefits and costs in EIA. EIA Review, 2(3):227-258.

Johnson, E.L. 1982. Risk assessment in an administrative agency. Amer. Statistician, 36(3):232-239.

Kelly, J. 1980. Wisconsin survey data. Complex Systems Group, Univ. New Hampshire, Durham.

Kunreuther, H. and J. Linnerooth. 1983. Risk analysis and decision processes: the siting of liquified natural gas facilities in four countries. Springer-Verlag, Berlin.

Kunreuther, H. and J. Linnerooth. 1984. Low probability accidents. Risk Analysis, 4(2):143-152.

Kunreuther, H. and J. Lathrop. 1981. Siting hazardous facilities: lessons from LNG. Risk Analysis, 1(4):289-302.

Linnerooth, J. 1982. Murdering statistical lives. In: The Value of Life and Safety, M. Jones-Lee (Ed.). Geneva, North Holland, pp. 229-261.

May, Wm.W. 1982. $'s for lives: ethical considerations in the use of cost/benefit analysis by For-Profit firms. Risk Analysis, 2(1):35-46.

McAllister, D. 1980. Evaluation in environmental planning. MIT Press, Cambridge, Mass.

Meyer, P.A. 1976. Obtaining dollar values for public recreation and preservation. Environment Canada, Vancouver, Tech. Report Series-PAC/T-75-6, pp. 12-15.

Mishan, E.J. 1982. Recent contributions to the literature of life evaluation: a critical assessment. In: The Value of Life and Safety, M. Jones-Lee (Ed.). Geneva, North Holland, pp. 81-94.

O'Hare, M. 1977. Not on my block you don't: facility siting and the strategic importance of compensation. Public Policy, 25(4):407-458.

Rowe, W.D. 1981. Methodology and myth. In: Risk Benefit Analysis in Water Resources Planning and Management, (Ed.) Y.Y. Haimes. Plenum, New York, pp. 59-89.

Schelling, T. 1968. The life you save may be your own. In: Problems in Public Expenditure Analysis, D. Chase (Ed.), Brookings Inst., Washington, D.C.

Schulze, W.D. and A.V. Kneese. 1981. Risk in benefit-cost analysis. Risk Analysis, 1(1):81-88.

Smith, R.S. 1979. Compensating wage differentials and public policy: a review. Industrial and Labour Relation Review, April, 32:339-352.

Smith, R.S. 1973. Compensating wage differentials and hazardous work. Office of Evaluation, U.S. Department of Labour, Washington, D.C. Technical Analysis Paper 5.

Sorensen, J., J. Soderstrom and S. Carnes. 1984. Sweet for the sour: incentives in environmental mediation. Environ. Mgmt., 8(4):287-294.

Starr, C. 1969. Social benefits versus technological risk. Science, 165:1232-1238.

Thaler, R. and S. Rosen. 1976. The value of life-saving. In: Household Production and Consumption, (Ed.) N.E. Terleckyj. NBER Press, Columbia University, New York.

Viscusi, W.K. 1976. Labour market valuations of life and limb: empirical evidence and policy implications. Public Policy, Summer, 26:359-386.

Viscusi, W.K. 1979. Employment hazards. Harvard Univ. Press, Cambridge, Mass.

CHAPTER 6
THE SOCIAL TREATMENT OF RISK

Peter Timmerman
Institute for Environmental Studies, University of Toronto

INTRODUCTION

When we look at the broader context within which risks are assessed and debated in society, we discover that many conflicts over risk are not conflicts directly over fact, but conflicts over the social context within which risks are identified, created, and evaluated. These conflicts may include significant clashes over what is to constitute appropriate evidence, what the domain and range of significant fact will be, and whether or not technical analysis is even applicable. By attempting to pre-empt this kind of conflict, some forms of risk assessment threaten to be seen as part of the problem, and not as part of the solution.

To examine this thorny issue, it is perhaps best to begin with the term "risk" itself. At the heart of the claims for "risk" is its claim to be useful as a bridging concept. It shares with a number of recent concepts (such as "vulnerability" or "resilience") the ability to act as a verbal pivot between the gathering of quantitative data about a system or series of events (real or putative) on the one hand, and their qualitative expression either in the entire system (or as interpreted by human observers assessing the overall situation) on the other. "Risk" is one such pivotal term: it gestures both in the direction of probability theory, statistics, etc., and also in the direction of personal threats, dangers, and uncertainties. Since it can be expressed both as a noun - the "risks" of this or that - and as a verb - to "risk" one's life - the question of what exactly one means when referring to "risk" is by no means simple; nor is it simplified by reference to "hazards" or "hazardousness", terms with ambiguities of their own. This kind of ambiguity can be used creatively or destructively, and it is obviously part of the increasing presence of "risk" in political and managerial parlance today, since these agents are being asked to cope with technical risks that people simultaneously perceive to be threatening personal hazards.

The "perception of risk" is the term which currently identifies what, more broadly, could be called "the social treatment of risk". Since it is widely used, however, it will be referred to in the first half of the rest of this chapter, where a survey is given of current thinking on approaches to this issue.

In the second half of this chapter, however, an alternative approach is proposed, based not so much on the **perception** of risk, as on the **conception** of risk. This may seem at the outset to be no more than a verbal quibble; but it will be argued that the latter approach - though still in its infancy - is actually more relevant to the social and political issues involved in risk assessment.

THE PERCEPTION OF RISK

HAZARD THEORY

As the term suggests, "perception" of risk invokes the empiricist model of the individual who perceives some object (usually visually) in the outside world. The supposed object impinges on the individual through a series of physical mechanisms (e.g. reflected photons, retinal rods and cones, and brain cells), resulting in a conscious "percept". As philosophers have noted, these perceptions may lead to various illusions and misapprehensions of what is actually transpiring in the world "out there", of which the most famous ancient example is the stick that appears to be bent because it is halfway under water, and which has now been replaced as an example by the much more complex and puzzling cases associated with the status of the observer in quantum physics. The scientific difficulty is in making or tracing the process which is supposed to match the real world "out there" with the perceiver "in here" (Hacking 1981).

The perception/illusion model has been brought into the risk debate through a number of pathways, but two are of particular interest: hazards theory, and the psychology of perception. In both cases - though the second has a longer history of technical experiment behind it - the aim is to come to some kind of understanding of why "ordinary people" suffer from illusions, misapprehensions, or act in seemingly irrational ways when confronted by the presence (or temporary absence) of various risks and hazards.

In the development of hazards theory, people's decisions with regard to various nature-related hazards provided the intial focus of attention. One facet of this was "White's Paradox", named after Gilbert F. White, the dean of hazards geographers and an expert in water management planning. The paradox denoted the unsettling (or perhaps resettling) discovery that the spate of dam building in the U.S. in the first half of the century created an increased incidence of certain types of flood disasters, contrary to expectations. The reason was simple: following the construction of dams, people moved into the hitherto dangerous or inaccessible floodplain, expecting to be protected by the dam (or bailed out by the government if they weren't). This was an example of an institutionally promoted set of hazards - or shift in the patterns of risk acceptance and avoidance - and it caused a certain amount of consternation (White *et al*. 1958).

The complex web of personal risk-taking, insurance, locational theory, and "hazards mapping" that was invoked by this phenomenon, and similar examples of it elsewhere (cottages along lake frontages, migration into California earthquake zones, farmers cultivating volcanic slopes, etc.) promoted the creation of hazards theory. This field of research borrowed from a number of disciplines, some already referred to, in order initially to describe, and latterly to explain people's experiences, strategies, and perceptions concerning the hazards and risks around them (Burton, Kates, and White 1978). The most substantial borrowing and overlap was with the similarly eclectic field of "environmental perception" (Whyte 1977); and through the 1970s there was a strong focus on the gathering of information through social surveys and various types of comparative analysis (White 1974). Some of these focussed on particular places ("The Hazardousness of a Place" was one title, Hewitt and Burton 1971); on particular types of events (e.g. climate related hazards such as drought) and on cross-sectoral approaches (Kates 1978).

The survey approach was undertaken by a number of practitioners. In the attitude survey, the process was designed to elucidate the "belief structure"

of the respondents, expressed usually through a number of salient components of that structure, which the survey attempts to discover, and quantify. In the most rigorous of these survey types (Fishbein and Ajzen 1975) the working hypothesis was that a person's behaviour is some function of his behavioural intention, and that this in turn depends upon a mix of factors, including attitude and other beliefs about the social context (McKennell 1970). Attitude is linearly linked to the set of salient beliefs to which the respondent adheres (as measured on various scales, e.g. the Likert 7-point scale). Critiques of this approach focussed on the criteria for salience; and subsequently other researchers attempted to apply factor or cluster analysis to more preliminary or more open forms of questionnaire.

Other approaches addressed the quite different responses internationally to various hazards, especially the differences between developing and developed countries in terms of impacts from hazardous events (Hewitt 1983). Part of this was undertaken through the survey approach (White 1974); but an anthropological approach was also developed, trying to articulate the relationship between various "risk strategies" adopted in various cultures and actual physical situations in varying uncertain environments (Thompson 1980; Laughlin and Brady 1978).

The question of hazardousness as a research topic became more and more involved with technical hazards in the middle and late 1970s, as a series of incidents involving uncertainties due to technical complexity took centre stage (e.g. Three Mile Island); and as a series of enquiries concerning the very large-scale impacts of certain technologies were carried out (e.g. the supersonic transport enquiry in the U.S.). Hazards theory had already been involved with examining the orientation of societies towards natural "low probability-high consequence" events, and the means by which societies adjusted in the short-term, or adapted in the long-term to the potential or actual consequences of such events. With the rise of concern over high technology hazards, the issue of the "dissonance" between the actual probabilities of certain hazards and the public perception of those probabilities became acute, and required a re-thinking (which is still going on) of what theory of human activity and explanation was to be used in order to make coherent sense out of the vast and disparate mass of hazard-related cases and data sets that had been gathered over the years. Here the overlap between the "environmental perception" side of hazards theory, and the psychology of perception was found to be useful.

THE PSYCHOLOGY OF RISK PERCEPTION

The patterns of expectation associated with the uncertainties concerning the expression of probabilities in everyday life, and in specialised areas such as gambling have been a subject for research (and profit) for centuries; but only in this century have these associations come under systematic scrutiny by, among other disciplines, perception psychology (and the related area of decision theory). There has, in fact, been a process of convergence, among decision theory, management theory, and economic psychology. Many patterns of expectation, or interpretations, of events are illusions generated by short-term pseudo-order in what, over the long-term, can now considered to be seen as randomness. Since people strive to seek some order, and thereby at least some interpretative control over the world, the hypothesis can be put forward that people form belief systems or guidelines in order to predict, stabilise, or sort out reality. These in turn influence perception, and can introduce misunderstandings or illusory notions of how reality is organized.

In those areas where people do not have access to statistical data and are unwilling or unable to carry out systematic analyses and comparisons, it has been argued (Kahneman et al. 1982) that they rely on "heuristics" - a set of general inferential rules. These are sometimes practical, consciously applied "rules of thumb"; but many of them are quite subtle, unconscious guidelines for making sense of a complex world. These are often of great use in simplifying, and in making rapid decisions under conditions of uncertainty; yet they can be systematically misleading at other times - prejudicing ("prejudging") incorrectly more rational outcomes.

Heuristics include (not exhaustively): availability, anchoring, and forms of bounded rationality.

Availability The ability to imagine and recall events is one way of heuristically relating the past to the present for the purpose of learning to avoid risks. Since frequent events are more "available" (Tversky and Kahneman 1974), this heuristic historically made adaptive sense in a natural environment within arm's reach. However, the provision of information and images through the modern media can increase the perceptual "availabilty" of certain risks, even though the risks are remote or rare. This can make people underestimate or overestimate the probabilities of various risky events, based on estimations related not necessarily to the frequency of the events, but on the frequency of mentions in the press.

It has been shown that the "availability"-related perceptions of risks among reasonably educated North Americans can be correlated with the frequency and characteristics of mentions of those risks in the press (Slovic et al. 1981). It is difficult to determine whether the press mentions those risks the public is concerned about, or whether the public is concerned about the risks the press mentions (there is probably mutual support); but there is little doubt that there is some skewing of perception going on, with the result that, for example, rare spectacular events are given a higher profile - "availability" - and hence given higher than appropriate probabilities of occurrence.

"Availability" is more complex than this, however. The more available certain risks actually are, the more difficult it becomes to determine what the public reaction will be. For instance, there is the well-known phenomenon of risk "denial" by those living in the shadow of certain risks (Kates 1962). In addition, the more people handle certain hazards, the less concerned they may become about them (Slovic et al. 1979). On the other hand, as the initial "availability" heuristic indicates, it is also possible that by talking a great deal about the risks associated with a certain rare event, the "availability" of that event increases. Slovic et al. (1979) note the situation of an engineer discussing all the possible variations of things that could go wrong, however remote, with a nuclear plant, in order to dismiss them: the effect on his audience may instead be that they were unaware that so many things could possibly go wrong!

Anchoring People tend to anchor themselves in the first strong perception of a type of event. This affects subsequent probability estimates. The phenomenon is easy to uncover. In one experiment by Tversky and Kahneman (1981), groups of students were presented with one of the two following number patterns, and asked to give the answer in 5 seconds. Either:

$$8 \times 7 \times 6 \times 5 \times 4 \times 3 \times 2 \times 1$$

or:

$$1 \times 2 \times 3 \times 4 \times 5 \times 6 \times 7 \times 8$$

The median estimate for the first sequence was 2,250, while the median estimate for the second was 512 (the correct answer being 40,320). Here, the "shot in the dark" computation is anchored in the earliest numbers.

Bounded Rationality The concept of "bounded rationality" originated in management studies and economic theory (Simon 1983), and refers to the fact that decisions are not - as theoretically required by standard approaches - optimizations of all data, but are in fact based on selected or most easily available data. The term "bounded rationality" can be expanded to refer to all of those heuristics that roughly (and often inaccurately) shape the simplifying of problems. As with anchoring, for instance, the ability to predict the behaviour of a simple system may be erroneously projected into an ability to predict the system as it becomes incrementally more and more complex (Fischhoff et al. 1982). Overconfidence has been directly studied (Fischhoff et al. 1978) in the drawing of confidence bounds around estimates, with the common result that even sophisticated analyists think they can estimate uncertain quantities with much greater precision than they actually can. Again, Tversky and Kahneman (1971) have shown that many experts have faulty intuitive expectations and understanding of statistics - they assume, for example, the representative nature of sample sizes that are actually too small to be representative. The low predictive power of many published studies in reputable journals has been repeatedly exposed. Slovic et al. (1980) tabulated the following common problems leading to the underestimation of risks by experts:

1. Failure to consider human error;
2. Overconfidence in current scientific knowledge;
3. Failure to appreciate "whole system functions";
4. Slowness in detecting chronic, cumulative effects;
5. Failure to anticipate human response to safety measures;
6. Failure to anticipate "common-mode" failures.

It is generally believed that these failures of experts are compensated for by their access to more and better information, and to a tradition of personal scepticism and coordinated group scepticism; but these compensations are very often only of real use in areas where the expert is already expert. Philosophers and analysts of "the science of science" have noted for years that an expert's scepticism is not universal, but is directed at certain foreground issues or "research programs" specific to each discipline (Lakatos 1970). Outside of that discipline, scepticism may be muted, misdirected, or it may ignore warning signals that would alert practioners on the inside. The relationship between what constitutes rational decision making, and the working agenda of scientists and experts is thus at least as complex as that between the rational citizen and his or her life context.

PREFERENCE TESTING

Given that perceptions of risk differ, and that attitudes to risk may shape people's reactions to the threat or existence of a variety of hazards, there has developed a methodological literature which attempts to cope with some of these issues in objective, data-gathering fashion. These can be roughly categorised as "revealed" and "expressed" preference testing, after Starr's terminology (1969).

The central notion of "revealed preference" is that the world around us represents the best available source of information on how people perceive and handle risk. That is, the history of the world can be seen as a series of

adaptations to various risks and hazards; and we have developed institutions and guidelines which - in varying degrees -already embody a myriad of decisions concerning risk. One familiar example is in the provision of insurance, which is based on actuarial data about risk. Note that this revealed preference is continually being updated. For example, in the insurance industry, although past policy pricing may be based on actuarial data and company histories, complex mechanisms have come into existence to gather and share risk information. Each company may specialise in an area (e.g. chemical industries, mining, athletics), while the industry as a whole maintains actuarial committees that evaluate mortality statistics. New relationships between risk information and policies are often mediated through the marketplace: when one specialist company starts offering lower policy premiums, thereby undercutting its competitors, this may be a signal that a new evaluation of the risk in that specialty has taken place, and the other companies will adjust their policy premiums accordingly.

The working assumption of this "revealed preference" approach is that the risks/benefits have already been worked out historically and that newer risks that are analogous or only slightly different from earlier risks can be assumed to generate similar values and preferences. There are a number of obvious flaws in this approach: history may have made a mistake (e.g. the exposure of volunteers to atomic bomb tests in the 1950s); the risk decisions made in the past may have been made on the basis of information now known to be faulty or superceded (e.g. the long story of asbestosis Meek 1988). Moreover, this approach, by definition, cannot hope to deal with quite new risks deriving from radical new technologies or chemicals.

The second, or "expressed preference" approach relies on social survey techniques, questionnaires, and other means by which people can be made to gauge or express their risk preferences and aversions. This approach is in the tradition of the "attitude survey" (see Fig. 1). These techniques are susceptible to many of the problems we have been sketching up to this point, problems which can be summarised as "decision-framing", i.e. the framework within which people make their decisions (or express their preferences based on their perceptions) is critically important in how those expressions get expressed. The way the questionnaire is presented, the questions posed, and the range of answers required are all factors that have to be taken into consideration in weighing the results. A difficult factor to deal with is the artificiality of any survey as compared to the actual response of people to real risks.

The techniques of "revealed" and "expressed" preference are similar to those found in the economics and welfare economics literature associated with cost/benefit analysis. This similarity is not co-incidental, since the measurement techniques for "risk" are often in the same objective, expressed utility model as technical neo-classical economics. The further presumption is that one might be able to marry both cost/benefit analysis and risk analysis into risk/benefit analysis. For the reasons outlined above, the success of such a process could only come in a very narrowly defined area. It is clear from the psychology of perception research - to go no farther - that people's subjective utilities can be quite contrary to an observer's expectation, and will thus falsify optimization patterns. There is, however, work currently underway in the economics literature on the theory of "rational expectations" which proposes to circumvent some of these problems. It is too early to say what effect this will have on risk analysis.

1 Home Gas Furnaces
2 Home Appliances
3 Home Power Tools
4 Microwave Ovens
5 Power Lawn Mowers
6 Handguns
7 Terrorism
8 Crime
9 Nerve Gas
10 Nuclear Weapons
11 National Defense
12 Warfare
13 Bicycles
14 Motorcycles
15 Motor Vehicles
16 Railroads
17 General Aviation
18 SST
19 Jumbo Jets
20 Commerical Aviation
21 Anesthetics
22 Vaccinations
23 Pregnancy, Childbirth
24 Open-Heart Surgery
25 Surgery
26 Radiation Therapy
27 Diagnostic X Rays
28 Alcoholic Beverages
29 Caffeine
30 Water Fluoridation
31 Food Coloring
32 Saccharin
33 Sodium Nitrite
34 Food Preservatives
35 Food Irradiation
36 Earth Orbit Satellite
37 Space Exploration
38 Lasers
39 Asbestos
40 Police Work
41 Firefighting
42 Christmas Tree Light
43 Cosmetics
44 Fluorescent Lights
45 Hair Dyes
46 Chem. Disinfectants
47 DNA Research
48 Liquid Natural Gas
49 Smoking
50 Tractors
51 Chem. Fertilizers
52 Herbicides
53 DDT
54 Pesticides
55 Aspirin
56 Marijuana
57 Heroin
58 Laetril
59 Amphetamines
60 Barbiturates
61 Darvon
62 Morphine
63 Oral Contraceptives
64 Valium
65 Antibiotics
66 Downhill Skiing
67 Fireworks
70 Football
71 Hunting
72 Jogging
73 Mountain Climbing
74 Mushroom Hunting
75 Recreational Boating
76 Roller Coasters
77 Scuba Diving
78 Skateboards
79 Sunbathing
80 Surfing
81 Swimming Pools
82 Fossil Elec. Power
83 Hydroelectric Power
84 Solar Electric Power
85 Non-Nuc. Elec. Power
86 Nuclear Power
87 Dynamite
88 Skyscrapers
89 Bridges
90 Dams

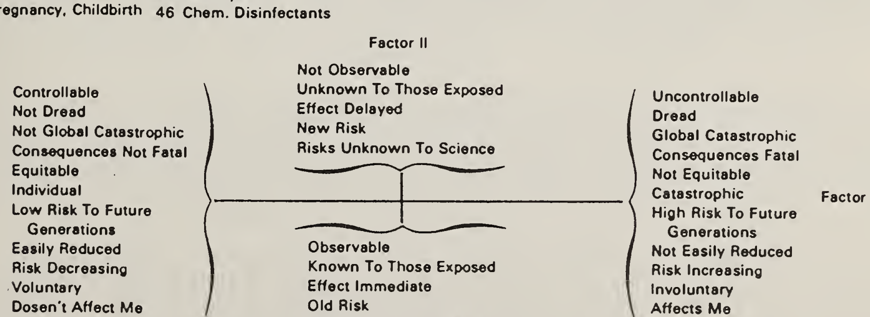

Fig. 1. Factors 1 and 2 of the three-dimensional structure derived from interrelationships among 8 risk characteristics in the extended study. Factor 3 (not shown) reflects the number of people exposed to the hazard. Source: Slovic et al. 1980

Be that as it may, Pushchak (1983) notes that the existing methodologies used in risk assessment are of three types, all of which incorporate the results of risk estimates calculated by risk analysis:

1. Cost-effectiveness analysis;
2. Weight-scoring methods;
3. Decision analysis.

Each of these methodologies incorporates some form of evaluation, either of effectiveness, or of the weighting of various risks by some process. If any of the material discovered in risk perception studies is to be incorporated here, it must either be quantitatively formulated, or be amenable to some kind of rigour, so that some form of commensurability is established - otherwise the process is comparing apples and oranges.

A deeper problem, identified by Wynne (1982) is that the whole framework for this kind of research may be systematically misleading, since it is based on attempting to persuade people to consider atomistically and precisely their wants and aversions. This is further based on the proposition - essentially political - that what is at stake in the process is the balancing-off of various expressible interests which are determinable as a specified goal or set of objectives. The more interesting question, of how people come to have those interests and wants, is not addressed. Quantitative surveys are likely to continue to give conflicting and counter-productive results, since:

> one of their (people's) central wants is to keep their options open and informal, to be free to revise and renegotiate their priorities and objectives in a social context that they find meaningful and open to their own influence (Wynne 1982).

In this perspective,

> Avoidance behaviour is no less rational than a goal directed one....Its function is to guarantee that the conditions exist in which (diverse) ends can be pursued (Wynne 1982).

The refusal of people to act in accordance with bureaucratic rationality is a key reason for "perception of risk" problems in the first place. But the above considerations suggest that different groups not only have different "risk strategies" by which they manage their lives; but that by trying to assimilate "risk" into one numerical model one can provoke the very conflict one is trying to resolve.

One alternative is to try and look at the problem of risk in a way that allows us to examine the use of risks as embedded in the contexts within which people actually operate, work, and live. Rather than ask: "What are the risks?"; we might instead ask: "What are people afraid of, and why?" We need to examine people's fundamental conceptions of risk.

It is easy to say that what we ought to do is to redefine risk, redirect the thrust of quantitative methodologies, and resolve the clashes between technical and public risk evaluations. One difficulty is that by cutting ourselves adrift from the quantitative, the ability to convince drops drastically, as does some measure of "quality control". Another difficulty is that there is, as yet, no convincing technique for grasping the qualitative - and this, after all, has always been something in favour of cost/benefit analysis: there is at least something concrete with which to work.

Yet, we can no longer go on as at present. The conflicts over risk do not seem to be lessening over time, nor is the dissonance between the quantitative and the qualitative dissolving into harmony. Simple pragmatism requires that we step back, and see if there is another alternative that can be brought to bear on this situation.

THE CONCEPTION OF RISK

In what follows, there is an attempt to develop the case for considering risk from a more "anthropolitical" perspective, or what has also been called the study of "political culture" (Wynne 1980; Thompson 1980) - an approach that tries to consider the conception of risks in the context of patterns of culture and social roles, and which tries to explain a broad range of characteristic responses to risk. To make this approach plausible, I have first put forward two case studies, both dealing with technical risks, and both rooted in contemporary society. The first case study is of the Mississauga Train Derailment in November 1979; and the second, the continuing attempt to locate a hazardous waste facility in the province of Ontario.

CASE STUDY 1: THE MISSISSAUGA DERAILMENT AND EMERGENCY

Just before midnight on November 10, 1979, a freight train carrying over a hundred cars derailed in Mississauga, Ontario, eventually requiring the evacuation of a quarter of a million people from the possible danger of a substantial release of chlorine gas out of one of the overturned cars. The Institute for Environmental Studies at the University of Toronto was asked to carry out a research project into the derailment. One task (carried out by the author with Ian Burton of the Institute) was to study and later reconstruct the higher level organisational response to the emergency. This involved the piecing together of an overall timetable of events, a series of key-actor interviews, and the use of transcripts of the meetings of the central Control Group of the emergency response (Burton _et_ _al_. 1981).

What became clear early on in the research was that, in order to understand the decisions made and the actions taken by the response organisations during the week of the derailment, it became necessary to see the organisational operations as falling into two different phases. These phases were not rigidly separated one from another, but one could demarcate them according to the players involved, according to the decisions that had to be taken, and interestingly enough, according to the ways risks were evaluated (or, indeed, deliberately left unevaluated).

In the first - most dangerous - "emergency" phase of the event, the response was handled by the emergency services: fire, police, and ambulance. These services were faced with a massive derailment, boiling liquid expanding vapour explosions (BLEVEs) which hurled tank cars of burning propane hundreds of yards in various unpredictable directions, and which threatened at any moment to break open a chlorine car, sending lethal gas rolling across the city of Mississauga. They were operating literally and figuratively in the dark concerning the risks involved, and the likelihood of their making it out of the emergency alive. What calculations they made were based on hunch, educated guesses, professional experience and the occasional back-of-the-envelope calculation.

The considered response of these organisations to this incredibly hazardous situation, filled with immense uncertainty, was to hurl themselves at it. There was no doubt that the people involved took what precautionary measures they could; but without being overly careless, they simply refused to evaluate their risks. There wasn't time; it wouldn't have done much good anyway; and besides, it was an "emergency" - this was their job.

If, to be crude about it, you had gone in at this point and asked these people - especially the senior officers - if they were aware of how much they were spending on all these services, the overtime, etc.; whether or not the costs would be recoverable; and whether they had consistently evaluated the risks and benefits each of the men involved in the emergency vis-a-vis the risks and benefits for the entire community - you would have been running a pretty severe personal risk yourself.

At such a time, one might have heard some variation on the standard refrain heard in emergency after emergency: "We'll worry about how much it costs after it's over." By the same token - although they would not have expressed it this way - one of the principles of "emergency" services is to plan to try and minimise the risks, but to be prepared to go into situations where the risks are "open-ended". Emergency services are designed (I exaggerate for emphasis) to worry about the risks retrospectively, after the situation is over.

After the initial emergency phase, which resulted in the evacuation of a quarter of a million people from a 25 km^2 area, Mississauga became largely a ghost-town, cut off from the world by police cordons. Inside the cordon, as train crews and chemical experts tried to seal up a leaking chlorine car, the response altered. It has been facetiously characterised by a remark that if the derailment had continued into a second week, they would have built an office building around it; but the essence of that remark is valid. From the transcripts, timetables, and interviews, it is clear that some of the most basic elements of an incipient bureaucracy (in the technical sense as defined by Weber 1946) were set in place.

There was, for a start, a large number of government agencies and political actors that had converged on the area, hiving off subgroups and personnel from their original positions and sending them to represent or assist in the derailment response. These agencies and respresentatives were oriented around a senior central Control Group, who found themselves having to limit access to their "cabinet meetings". The meetings themselves - for which transcripts began to be made, in order to ensure accuracy for the historical record - took on more and more of a hierarchical, managerial, function as the week progressed, as it became necessary for a number of reasons to review the actions and competence of various hitherto independent emergency workers.

In this context, as time passed and as the need to present an image of competent, consistent, handling of the event became a very important factor in keeping the confidence of the public (especially considering the continuing pressure to allow evacuees to return home), the Control Group began to concern itself with "risk" in the ways enshrined in the models of rational risk assessment. First of all, there were substantial risk analyses done and canvassed, using all available expertise and data; second, these risk analyses were factored directly into decisions about areal extent of possible future risks; third, the question of the methodology and comparison of risk surfaced in various contexts; and fourth, the issue of the public perception of risk as compared to that of the experts came significantly to the foreground.

Now it could very well be argued that, in this phase of the response, the carrying out of risk assessments was as much as anything one of the virtues of having time to draw one's breath, of being able to consider without distraction and with full information and expertise the range of risks involved. This is absolutely true, but the further truth is that the matching of this type of risk assessment to the rational risk assessments of the models is not coincidental: that is, when we can sit at our desks and draw breath, and consider abstractly the various ramifications of the problem in a logical and consistent fashion, disregarding the constraints of time, the uncertainties of the immediate context, and personal hazard, then technical or instrumental risk analysis may be appropriate.

In an emergency, such an analysis is not only inappropriate, but it can be very dangerous, in the same way that a fencer who stops to think about what he is doing is liable to be skewered by his opponent. Something else is also at work, however. An "emergency" is an example of what philosophers call a "performative utterance" (Austin 1962); that is, just as saying "I dub thee knight" or "I now pronounce you man and wife" does something in the very act of speaking, so calling something an "emergency" invokes a context in which such things as costs fly out the window. Once, for example, one is accepted into an emergency ward, society has determined that the sky is the limit on costs, since, in this specific context, the cost of saving a human life is deliberately ignored. A similar "cost-free" ritual occurs in transplant cases: although we are well aware that society is always surreptitiously putting costs on human lives by setting priorities, probabilitistically-based guidelines for safety, etc., when it comes to specific cases we ritually refuse to do so. The same is true in various kinds of emergencies. The most extraordinary of these rituals goes well beyond simply ignoring costs, and involves deliberately incurring risks for the sake of the sanctity of dead bodies - for instance, in the case of rescue workers going up on treacherous hillsides to recover remnants of bodies scattered after plane crashes.

Organizations such as the police and fire departments have a particularly complex set of relationships to the calculating or ignoring of risk. In an emergency, a fire chief has to take into account the physical risks as best he can evaluate them, he has to consider the duty he and his men have to take on very large short-term risks so that other people (the public) do not have to take either short or long-term risks, and he has to take into account the morale of his individual men and squads, which often means silently allowing them to be courageous and fearless in the face of serious danger.

This intricate interplay of context, role, responsibility, and sometimes personal worthiness is one aspect of what I would call the "conception of risk", and it goes well beyond discussions of whether or not people empirically "perceive" various risks. In one sense, it is a question of one's attitude towards risk and of the cultural context of risk; but that is far too simplistic a way of putting it: it is more a question of how uncertainty is woven into aspects of our lives.

CASE STUDY NO. 2: NOT-IN-MY-BACKYARD SYNDROME

The second case study - which can be presented in a more abbreviated fashion - provides a different perspective on the same issue.

The "Not-In-My-Backyard" syndrome - as it has been dubbed - is at the heart of the continuing dilemma about the location of new hazardous waste treatment facilities (in this instance, in Southern Ontario). An early attempt

to locate such a facility was undertaken by the standard political and bureaucratic agencies of the Ontario government, but the public opposition was so strong that it was decided to create an Ontario Waste Management Corporation (OWMC) as an alternative approach to a solution.

This corporation's mandate and raison d'etre was to find a technical and bureaucratic solution to the facility siting problem by, among other things, consideration of the entire spectrum of hazardous waste management issues in the province. As might be expected, this resulted in the production of a number of technical studies, followed by a series of rationalisations and syntheses, all of which culminated in 1984 with the selection of a number of "optimal" sites according to OWMC criteria.

It seems to have been an assumption of OWMC's that, although their "selling job" would be difficult, they would ultimately prevail, since people would be convinced of the case for one of these sites on its merits as canvassed technically, thoroughly, and with unimpeachable integrity by OWMC. Yet, in spite of the standard call for "public input", the provision of forums for carefully structured individual presentations, expert reviews of the literature on the "perception of risk", and information "show-and-tell" sessions across the province, the public in the areas chosen for consideration are strongly opposed to having the proposed project in their area. Indeed, rather than having to cope, as earlier, with the outcry against one site chosen by the existing system, OWMC created an outcry in each of the candidate areas it selected under its new approach. At the time of writing (Summer 1989) hearings on the environmental impact statement (EIS) are planned for the Fall of 1989, the project will face strong opposition, and it is possible that legal or further recourse to political decision-making will have to be undertaken in order to site the facility. OWMC's work will thus become a rationale for political action, rather than a basis for reasoned agreement.

A cynic might suggest that, considering the already polarised situation in Ontario, this was inevitable from the outset; but it might also be argued that OWMC, like everyone else involved with hazardous waste management, has just come up against the "perception of risk" issue in a particularly acute fashion.

A maddening version of this occurs in miniature during hazardous waste incidents where the public and various interest groups are deeply involved. There will be - at a public meeting perhaps - a presentation of technical information, hazard maps, protocols for experimental analysis, etc., of the problem. These will be attacked by the public on technical grounds, focussing on the uncertainties of the techniques involved, the measurements of low-level risk, and the lack of absolute truth associated with any scientific venture. If these attacks fail - and often they do not - then there is a retreat to the next line of defence: "Even if the studies show that there is only a marginal risk, we still don't want the facility here".

It is at this point that the word "irrational" starts getting thrown about; and rather than call the public (taxpayers and voters) irrational, refuge is taken in the diplomatic term "perception of risk". As we have indicated, this is only a solution if one is considering the situation from the managerial perspective; and, from that perspective, it appears to be a simple step to such further considerations as perhaps compensating people for "perceived risks" which the experts and managers involved are convinced are illusory. This may solve the political problem (although this is doubtful), but it clearly does not serve as a point around which consensus can be built,

since there is no consensus - not just about the risks, but about what role "risk" is to play in the given situation.

SOCIAL RATIONALITIES

Two disciplines - philosophy and anthropology - have been concerned with the content and the context of dispute; and in what follows, insights from a philosopher and an anthropologist are presented as the beginnings of a potential methodology for approaching a more broadly based "conception of risk".

The hazardous waste dilemma is an example of a moral as well as a technical conflict, and one of the characteristics of moral dilemmas in our time is that they are often unresolvable, because both sides in the conflict appeal either to competing fundamental principles, or, in lieu of those, to justification by sincerity (methodological or emotional). In the hazardous waste dilemma, the proponent side, as an arm - however remote - of the government, is invoking the utilitarian principle of the greatest good for the greatest possible number, and is also calling upon citizens to sacrifice a certain amount of their personal well-being for the good of the community as a whole. On the other side, embattled citizens and their families, communities, and spokespeople, are invoking a range of principles, including what have been called "trump rights", i.e. their absolute rights as citizens not to be treated as means to someone else's end (Dworkin 1978).

This kind of fundamental clash has been recently characterised by the philosopher MacIntyre (1981). MacIntyre asks us to imagine a society in which there are four or five moral systems that, by some catastrophe, are cut adrift from their original context or roots. By a process of cultural amnesia, the contexts within which these systems functioned smoothly have been lost, and the fragments of what remain are incompatible or worse: they continually talk past each other, or misinterpret each other. Each system appears to be valid, and conclusions follow validly from the initial premises - the only problem is that there is no acceptable method of giving one system priority over another. MacIntyre suggests that our world can be seen as the world that has suffered such an amnesiac catastrophe; that is, modern conflict can be seen as the result of the vacuum created by the removal of previous standard, acceptable, publicly validatable models of what constitutes right, proper, and virtuous conduct.

MacIntyre's more subtle point is that not only do we find different people hanging onto - or subscribing to - different fragments of moral systems, but we also find ourselves, as individuals, continually oscillating between different fragments ourselves. In addition, one of the strongest cultural elements of modernism is our ability to shift quite easily from role to role and fragment to fragment in different areas of our own lives, due to the superficially homogenising nature of our language and the diversity of our society. We do not, for example, operate by the same set of rules at home as we do at work: we do not treat our families in bureaucratic or utilitarian ways. When we do treat them in this fashion, this is usually an indication that something has gone seriously wrong in our domestic lives.

In the risk context, we can see some of the same flexibility combined with context-dependence at work. The hazardous waste dilemma already described is a classic example of the ability of the concerned citizen to:

(a) recognise the power of scientific technology as an explanatory device;
(b) criticise it on its own terms;
(c) yet ultimately appeal to a different context, based on what is appropriate in order to defend one's family from external threat.

To put it at its strongest: utilitarian risk assessment can be seen as both inappropriate and even immoral, when seen from the perspective (say) of a father, having to make a decision regarding his own children. It is a fundamental attack on the the role of parenthood to accept even the slightest increase in risk on behalf of your children for general utilitarian considerations, or for monetary compensation. In public meetings, these kind of appeals to moral absolutes appear side by side with arguments over technical details.

The bewilderment occasioned by these mixed signals is considerable, since observers (or "risk managers") can understand the language and the claims of those potentially affected, but the basis of these claims keeps alternating in ways that can sometimes be fathomed, sometimes not. So that, for example, as a risk manager, one may be presented with a technical rationale for some level of acceptable risk; but one knows at the same time, that it will be politically unacceptable. This is particularly true in areas of serious conflict (like hazardous waste management, where personal health may be at risk), since it is in these contexts that we may - for the first time - see the field strength of some of our basic beliefs, since, (to continue the metaphor) it is at this time that the situation has become polarised.

As Wynne (1982) pointed out, this can be seen as a clash of rationalities, as well as a clash of moralities. Hollis (1979), for example, describes his experience during a severe British drought in the mid-1970s, when the government was exhorting the citizenry to conserve water. Looked at in terms of individual rationality, Hollis should have used up all the water he needed; but in terms of the "good of the community", he would be unreasonable to do so. The government was appealing to people to be reasonable, but this went against the rational utilitarianism of most of what the government was saying in other contexts. This "reasonable/rational split" brought out the clash of two competing types of rationality: utilitarian, context independent, and instrumentally "rational", as opposed to more context-dependent, communitarian "reasonableness". This latter kind of rationality is one with vague contours and open boundaries, but which obviously exists: it is invoked, for example, by juries that under common law are required to take into account what a reasonable person, under conditions of uncertainty, ought to have done in a real situation. Another version of this rationality is associated with craftsmanship: mechanical rules and blueprints are no substitute for the evolving relationship between (for example) a master carver and the wood he is constantly reshaping, revaluating, and exploring according to an intuitive rationality and logic based on the given character of the object (Oakeshott 1950).

This clash of perspectives and rationalities is reminiscent of the continuing debate in anthropology over the possibility of interpreting one culture's outlook on the universe to another culture, or one tribe to another. This is particularly related to the work of Mary Douglas, whose anthropological researches (1970, 1975, 1978) have been into patterns of "social accountability" in a number of tribal systems. These patterns, expressed by and reinforced in religious cosmologies, political structures, and moral taboos,

are ways in which tribes cope with misfortune (i.e. risks) and events or objects (i.e. pollutions).

The link between her work and MacIntyre's, and that which has operational significance, is that Douglas has attempted to develop methods of mapping the various "field strengths" of the risk aversions and acceptances as embodied in the cultural contexts of tribes - including the tribe known as "Western civilisation". In her most recent work, "Risk Acceptability According to Social Sciences" (1985) she suggests that one of the virtues of an anthropological approach is that it "combines an analysis of rational behaviour with an account of the ethical constructs which are used for focussing social issues". Following on from her work, anthropologists and social scientists such as Thompson (1982) and Cotgrove (1982) have applied this mapping process to current tribal battles in our world - e.g. the energy debate. In this way, the arguments over risk can be discussed and evaluated, in terms that relate the ethical foundations of the conflicts to their contextual universes.

In their paper proposing the creation of a cultural theory of risk, Thompson and Wildavsky (1982) adapt a mapping model of Douglas' which originally sought to convey various aspects of a theory of the patterning of culture. (see Fig. 2). The basis of this theory was that what constitute cultural patterns or social biases are neither objective facts per se (i.e. unconstrained empiricism), nor subjectivities (i.e. unconstrained relativism), but a combination of the two (i.e. constrained relativism) mediated through social pressures and the interaction of individuals, and subsequently expressed in different cosmologies and attitudes towards - among other things - risk.

The four quadrants in the diagram relate to various strengths of the underlying dynamics of the social pressures involved. The horizontal line (left to right) moves from maximum individual imposition of power to maximum group imposition of power; while the vertical line (bottom to top) moves from maximum individual imposition of "grid" or meaningful categorisation on an uncertain world, to maximum group imposition of grid on the world. The mutual interactions of these grids and groups create and support systems of meaning, which enable people to stabilise and categorise their views of the world; and to incorporate (or reject) certain hazards, threats, and anomalies.

The important methodological point is that, by this approach, the cultural attitudes towards risk can begin to be mapped, since cultures have regularities, "forms of life", accessible rituals; and many of these are "risk strategies" by which people shape and protect their lives in the face of the fundamental uncertainty that life must be lived forward through historical time.

If there is any validity to this approach, it provides a new tool for the mediator in risk conflicts. For, in a sense, the mediator can be seen as a type of anthropologist trying to explain the fundamental notions of one tribe to another (and perhaps to himself or herself as well), thereby trying to lay bare the presuppositions of each side, and the differences that must somehow be addressed. This is done in the first instance by trying to describe the unfolding situation to all sides in a mutually comprehensible interpretative language - which was, after all, the aim of the more limited notion of risk with which analysts began.

1. Only short term risks perceived with any clarity.
2. No options as to acceptance or rejection. Risk acceptance a "fact of life".
3. No social boundaries to maintain so little concern for pollution and therefore few, if any, nonexistent risks.
4. Personal rewards seen as resulting from luck not personal risk taking and they are not resented.

1. Short and long time risk perceived. Long term seen as different from short term but controllable.
2. Risk averting. If risks cannot be avoided completely they are spread.
3. Many boundaries and distinctions to be maintained so elaborate pollution concepts and high incidence of nonexistent risks.
4. Personal risk for personal gain penalized. Personal risk for totality's gain rewarded.

Grid

+

"Life is like a lottery" (strongly negative group, strongly positive grid)

Caste-ist (strongly positive group, strongly positive grid)

- **Hermit*** (low group, low grid) Group +

Entrepreneur (strongly negative group, strongly negative grid)

-

Sectist (strongly positive group, strongly negative grid)

1. Long and short terms perceived. Dominance of short term over long term maintains expansive optimism.
2. Risk as opportunity.
3. Little concern for pollution (entrepreneurs profit from removal of social boundaries)
4. Personal risk for personal reward approved (even if it causes coercion). Personal risk for benefit of totality less popular.

1. Short and long terms perceived. Concern for survival causes long term to dominate short term.
2. Strong aversion to all risks except those involved in the defense of wall of virtue.
3. Pollution concerns all clustered around a single social boundary and give rise to many nonexistent risks.
4. Zero-sum mentality penalizes the personal risks for personal gain. Only risk taken for the totality are rewarded (often posthumously).

*

1. Only short-term risks perceived with any clarity.
2. If their avoidance would require social involvement (e.g. risk spreading through reciprocity and social obligations) then acceptance is lesser of the two evils.
3. Little concern for pollution so few, if any, nonexistent risk.
4. Personal risk taking for personal reward approved of as long as it does not involve coercion of others.

Fig. 2. Grid/group pattern Source: Thompson and Wildavsky 1982

CONCLUSIONS

The debate over risk is a search for some common ground, but it is bedevilled by the fact that uncertainty, and decisions over risk play a very large part in people's sense of shaping their own lives. Until we begin to grapple with the differences entailed by the intricate social and moral conflicts generated by aspects of risk, risk assessors are unlikely to escape continuing public paralysis and disagreement, since the currently assumed risk methodology is the expression of one "tribal bias", that is, the quantitative, utilitarian, bureaucratically rational approach to the solution of problems. There is nothing automatically wrong with this approach, except that its unwarranted extension provokes considerable hostility in a polarised debate, since it is seen, not as neutral and objective, but as a further extension of the one sided force which provoked the conflict in the first instance. In this way, unless risk assessors and environmental managers are careful, they are likely to be seen to be part of the problem more often than they are part of the solution.

Part of the problem associated with the social treatment of risk in EIA is that the current impasse derives from the historical trends out of which not only the rise of technological risks on a large scale developed, but also the development of the techniques associated with "risk analysis" and "risk assessment". One of the aims of this chapter has been to point out that valuable as some of these techniques are, they are not sufficiently subtle to handle social conflicts over risk, and that, in some cases, they are actually in the way. While some of the innovative alternatives described above certainly have their problems as well, they can make some claim to be grappling with the real complexities of the social dynamics involved in current conflicts over risk. For this reason alone, they deserve consideration in EIA.

REFERENCES

Austin, J. 1962. How to do things with words. University Press, Oxford.
Burton, I., M. Kliman, D. Powell, K, Schmidt, P. Timmerman, P. Victor, A. Whyte and J. Wojick. 1981. The Mississauga evacuation: final report. Ministry of the Solicitor General, Ontario.
Burton, I., R.W. Kates and G.F. White. 1978. The environment as hazard. Oxford University Press, New York.
Canadian Environmental Assessment Research Council (CEARC). 1986. Philosophy and themes for research. CEARC, Ottawa.
Cotgrove, S. 1982. Catastrophe or cornucopia. Wiley and Sons, New York.
Douglas, M. 1970. Natural symbols. Penguin Books Ltd., Harmondsworth.
Douglas, M. 1975. Implicit meanings. Routledge and Kegan Paul, London.
Douglas, M. 1978. Cultural bias. Occasional Paper No. 34 of the Royal Anthropological Institute of Great Britain and Ireland, London.
Douglas, M. 1985. Risk acceptability in the social sciences. Russell Sage Foundation, New York, N.Y.
Dworkin, R. 1979. Taking rights seriously. Harvard University Press, Cambridge, Mass.
Fischhoff, B., P. Slovic, S. Lichtenstein, S. Reads and B. Coombs. 1978. How safe is safe enough?: A psychometric study. Policy Sciences, 8:127-152.
Fischhoff, B., P. Slovic and S. Lichtenstein 1982. Lay foibles and expert fables in judgments about risk. The American Statistician, Part 2, 36(3):240-256.
Fishbein, M. and I. Ajzen 1975. Belief, attitude, intention and behaviour: an introduction to theory and research. Addison-Wesley, Reading, Mass.

Grima, A.P., P. Timmerman, C.D. Fowle and P.H. Byer. 1986. Risk management and EIA: research needs and opportunities. Canadian Environmental Assessment Research Council, Ottawa.

Hacking, I. 1981. Scientific revolutions. University Press, Oxford.

Hewitt, K. 1983. Interpretations of calamity. Allen and Unwin, London.

Hewitt, K. and I. Burton 1971. The hazardousness of a place. Department of Geography, University of Toronto, Toronto.

Hollis, M. 1979. Rational man and social science. Rational Action, (Ed.) R. Harrison. University Press, Cambridge, England.

Kahneman, D., P. Slovic and A. Tversky. 1982. Judgement under uncertainty: heuristics and biases. Cambridge University Press, New York.

Kates, R.W, 1962. Hazard and choice perception in flood plain management. University of Chicago, Department of Geography, Research Paper No. 78, Chicago.

Kates, R.W. 1978. Risk assessment of environmental hazard. SCOPE 8. Wiley and Sons, New York.

Lakatos, I. 1970. Falsification and the methodology of research programmes. Criticism and the Growth of Knowledge, (Eds.) I. Lakatos and A. Musgrave. University Press, Cambridge.

Laughlin, C.D. and I.A. Brady (Eds.). 1978. Extinction and survival in human populations. Columbia University Press, New York.

MacIntyre, A. 1981. After virtue. University Press, Notre Dame, Indiana.

McKennell, A.C. 1970. Attitude measurement: the use of coefficient alpha with cluster or factor analysis. Sociology, 4(2):227-245.

Meek, M.E. 1988. Health effects of asbestos. In: Information Needs for Risk Assessment, (Eds.) C.D. Fowle, A.P. Grima and R.E. Munn. Institute for Environmental Studies, University of Toronto, EM-8:239-260.

Oakeshott, M. 1950. Rational conduct. Rationalism in Politics and Other Essays (1962). Methuen and Co. Ltd., London.

Pushchak, R. 1983. Risk assessment in the siting of hazardous waste facilities: methods and experience. Ontario Waste Management Corporation, Toronto, Ontario.

Simon, H.A. 1983. Reason in human affairs. Stanford University Press, Stanford, Calif.

Slovic, P., S. Lichtenstein, and B. Fischhoff 1979. Images of disaster: perception and acceptance of risks from nuclear power. Energy Risk Management, (Eds.) G. T. Goodman and W.D. Rowe. Academic Press, London.

Slovic, P., B. Fischhoff, and S. Lichtenstein 1980. Facts versus fears: understanding perceived risk. Societal Risk Assessment: How Safe is Safe Enough? Plenum Press, New York.

Slovic, P., B. Fischhoff, and S. Lichtenstein 1981. Perceived risk: psychological factors and social implications. Proceedings of the Royal Society of London. A-376:17-34.

Starr, C. 1969. Social benefit versus technological risk. Science, 165:1232-1238.

Thompson, M. 1980. Political culture: an introduction. Intern. Institute for Applied Systems Analysis, Laxenburg, Austria, WP-80-175.

Thompson, M. 1982. Among the energy tribes: the anthropology of the current policy debate. International Institute for Applied Systems Analysis, WP-82-59, Laxenburg, Austria.

Thompson, M. and A. Wildavsky 1982. A proposal to create a cultural theory of risk. The Risk Analysis Controversy, (Eds.) H.C. Kunreuther and E.V. Ley, Springer-Verlag, Berlin.

Timmerman, P. 1984. Ethics and hazardous waste facility siting. Institute for Environmental Studies and Department of Civil Engineering, WM-84-11, University of Toronto, Toronto, Ontario.

Tversky, A. and D. Kahneman. 1971. The belief in the law of small numbers. Psychological Bulletin, 76:105-110.

Tversky, A. and D. Kahneman 1974. Judgment under uncertainty: heuristics and biases. Science, 185:1124-1131.

Tversky, A. and D. Kahneman. 1981. The framing of decisions and the psychology of choice. Science, 211:453-458.

United States Environmental Protection Agency (US EPA). 1984. Risk assessment and management: framework for decision making. U.S. EPA.

Weber, M. 1946. From Max Weber: essays in sociology, (Eds.) H.H. Gerth and C. W. Mills. Oxford University Press, New York.

White, G.F. 1974. Natural hazards: local, national and global. Oxford University Press, New York.

White, G.F. et al. 1958. Changes in urban occupance of flood plains in the United States. Department of Geography, University of Chicago, Chicago. Research Paper No. 57, Chicago.

Whyte, A.V. 1977. Guidelines for field studies in environmental perception. MAB Technical Notes 5, Paris UNESCO.

Whyte, A.V. and I. Burton (Eds.). 1980. Environmental risk assessment. Wiley and Sons, New York.

Whyte, A.V. and I. Burton 1982. Perception of risks in Canada. Living with Risk: Environmental Risk Management in Canada, (Eds.) I. Burton, C.D. Fowle, and R.S. McCullough, IES, Univ. Toronto, EM-3:39-69.

Wynne, B. 1980. Technology, risk and participation: on the social treatment of uncertainty. Society, Technology, and Risk Assessment, (ed.) J. Conrad. Academic Press, 1980.

Wynne, B. 1982. Institutional mythologies and dual societies in the management of risk. The Risk Analysis Controversy, (Eds.) H.C. Kunreuther and E.V. Ley. Springer-Verlag, Berlin.

CHAPTER 7
ENVIRONMENTAL RISK ASSESSMENT AND COMMUNITY IMPACT MITIGATION

A.P. Grima
Institute for Environmental Studies, University of Toronto

INTRODUCTION

What is a reasonable and acceptable risk? In a society where technologies are becoming increasingly complex, it is reasonable to expect that the legal and administrative procedures for assessing environmental risk will also become more elaborate. The challenge - and opportunity - of legal and administrative procedures, such as environmental impact assessment (EIA) is to avoid costly mistakes, mitigate unwanted consequences and achieve a sense of equity to affected communities and individuals. In order to meet the challenge and maintain its credibility, EIA needs to expand its theoretical and operational framework to include risk assessment more explicitly and to take into account the implications of including risk assessment in its scope of inquiry.

EIA is one of many bases for decisions that allocate natural resources and environmental quality (such as clean air or water or specific sites for hazardous facilities). Such allocative decisions are difficult for four reasons. First, there is considerable scientific uncertainty surrounding the consequences of projects such as drilling for oil, the operation and decommissioning of nuclear power plants and the disposal of hazardous wastes. Second, natural resources and environmental quality are scarce in the economic sense, i.e. they are not available in unlimited quantities at no cost. Third, there are conflicts among the users of these resources. Fourth, such decisions may result in substantial risk or hazard.

Environmental impact assessment and risk analysis (RA) may be considered as "planning" instruments designed to improve environmental and natural resource decisions by improving the scientific and technological basis for such decisions. In theory, the public interest is best served by a decision-making process that reflects the best scientific understanding about environmental decisions and lets the political process deal with the issues of scarcity and conflicting interests by requiring elected politicians to make final dispositions either in their capacity as members of the cabinet or as law-makers in parliament. The judicial system also plays an important role; in a liberal-democratic-capitalist system the judiciary interprets and implements the stated intentions of the elected law-makers. In reality, both EIA and RA are implemented within a context where certain interests are more powerful than others and where scarcity is resolved in such a way that reflects current relative prices and income distributions.In other words, EIA and RA already deal implicitly with scarcity and conflict and, often enough, these issues are explicitly aired in the public fora; indeed conflict and scarcity tend to attract more public attention than the scientific discussion on risk and uncertainty which occupies most of the time and effort in EIA. The application of EIA to projects that involve risk exacerbates this problem because events characterized by low probability with serious consequences (e.g. an accident in a nuclear power plant or oil well blow out) tend to be highly controversial; they also exhibit a high level of scientific uncertainty as opposed to more frequent events such as car accidents.

The challenge is to treat EIA and RA more as decision-making processes than planning exercises and to explicitly include consideration of scarcity and conflict. This chapter will not cover the issue of scarcity in EIA and RA; there is an extensive literature on demand management, pricing mechanisms, effluent and emission charges, transferable discharge permits and compulsory insurance to cover third party damages (see Griffith et al. 1981 for a bibliography). These policy instruments could be examined in EIA to make more costeffective (i.e. more efficient) decisions and therefore to make the best use of scarce resources.

This chapter deals with the challenge of addressing conflicting interests in the EIA process and particularly the difficulties that the inclusion of risk analysis presents in this regard. It is a challenge that requires attention not only because special interest groups (or special value groups, if we wish to be more polite) constantly keep conflict and equity in the public eye but, more importantly, because the credibility of EIA will depend on whether conflicting interests are more explicitly examined rather than ignored to surface again as emotionally-charged criticisms of the technical competence of the EIA. The Spadina Expressway, Pickering Airport and South Cayuga episodes, in Ontario, are cases in point: in each case the technical basis for the decision outcome was so forcefully challenged that the original decision was reversed. The reversal was largely based on political grounds but under adversarial questioning the scientific and technical basis of the original decision was badly dented too.

In other words, the explicit consideration of conflicting interests provides the opportunity to critique the scientific and technical basis for the decision outcome from alternative perspectives sooner rather than later. The reason is that many of the environmental risks that require assessment and decisions are characterized by:

1. impacts on groups who have to bear the risk of unwanted and unwilled, outcomes more heavily while society at large reaps the benefits;
2. uncertainty about outcomes and probabilities but certainty that some risk exists.

Impacted groups and communities resist these decisions: "not in my back yard" (NIMBY), they argue. The NIMBY syndrome is a direct result of these two characteristics in reaching what Fischhoff et al. (1981) call acceptable risk assessment and decisions. Whereas the politicians, administrators and scientific community need to ask "How Safe is Safe Enough?", the community at risk asks "How Risky is Too Risky?"

Ideally an administrative-regulatory regime (such as EIA) would be expected to accomplish three tasks:

1. it would produce decisions that are considered **competent** (i.e. well-founded and effective) by experts in the field (e.g. biophysical scientists, economists);
2. it would produce **fair** decisions;
3. it would allow as much consultation with and participation by affected parties as possible;
4. it would accomplish goals at least cost (i.e. be cost-effective).

The rest of the chapter is divided into three parts, roughly corresponding to the first three tasks listed above. As noted, the issue of cost effectiveness is not addressed in this chapter, except to point out that more cost-effective, conservation-directed strategies should be considered in environmental decisions (see Grima 1972, and Paine and Grima 1984 for two empirical applications of cost-effective strategies in urban water management and SO_2 abatement respectively). The next section focusses on the role of science and rationality in environmental decisions, i.e. on "acceptable risk" decisions that fall within the strict, formal rational approach to decision making. Competence follows logically from the understanding of technical experts who use formal analysis or their best professional judgement. The risk in novel technologies is inherent in the high degree of uncertainty of outcomes and this inherent risk, therefore, changes the connotation of competence in doing EIA. In the brief section on equity, the thesis is put forward that the economics of risk-benefit analysis depend on the choice of a suitable system of ethics; the case for compensated risks is introduced in this section. The section on consultation and participation focusses on some innovative approaches to community participation in the decisions and in sharing the benefits of risk-bearing projects. The aim of the chapter is to outline an expanded epistemological framework for EIA that needs to include an assessment of risk and to argue that the inclusion of risk analysis has important implications for EIA, particularly for community choice and community impact mitigation.

SCIENCE AND RATIONALITY IN ENVIRONMENTAL DECISONS: NEED AND LIMITATIONS

Few would disagree with the basic principle that each EIA should reflect the best available expert advice and that the recommendations and the analysis leading to the recommendations have to be recognized by the experts in the field to be competent. Competence requires that the investigation conforms to recognized canons of scientific and engineering principles and empirical research. While forecasting of any impacts is always uncertain, the argument here focusses on the scientific uncertainty on which the forecasting of impacts is based. To take two extreme examples, forecasting impacts based on the law of gravity are more certain than forecasting the impacts of a nuclear reactor meltdown. In the remainder of this section, I first discuss the complications that arise when the investigation yields results that are uncertain and then I discuss the rational approaches to decision-making. Although the emphasis is on the limitations of these approaches, this discussion is not meant to replace the rational or scientific approach with something else; science and rational discourse are essential to reaching competent, credible decisions. The critique is meant to provide directions for an expanded framework for making improved analyses, especially EIAs in those instances where acceptable risk looms large in the analysis.

SCIENTIFIC UNCERTAINTY

There are numerous occasions when the science behind decisions or recommended courses of action is not questioned, because the scientific understanding is widely shared (Grima *et al.* 1986; Fowle 1986). For example, the science and technology of conventional wastewater treatment is well understood. The science becomes less than uniformly accepted when

it goes beyond the theoretical relationships and fitted equations (i.e. beyond positivistic science, narrowly defined). In the environmental field, relating effects to causes may not be straightforward because empirical evidence and statistical correlations may include several collinear variables. For example, low levels of pH are correlated to forest damage but the increase in SO_2 levels is only one of several stresses on the terrestrial ecosystem. The line dividing informed speculation from scientific understanding may be very thin (cf. the climate change issue).

This type of scientific controversy is particularly pertinent to EIA since the conclusions and recommendations based on "controversial" science and technology have to be presented as a consensus rather than as a universally accepted finding. The inclusion of risk analysis in EIA - and many EIAs are about acceptable risk decisions - calls for recognition of the limitations of data availability and differences in methodology. For example in dose-response curves, one has to extrapolate from animal studies in laboratories or from epidemiological (i.e. statistical correlation) studies or from exposure to high dosages in accidents. The implications of this aspect of risk analysis for EIA is very clear: the "limitations" or "qualifications" of the scientific evidence - what Lowrance (1976: 56-70) calls "problems of inference" - should be made explicit and emphasis be put on the mitigative procedure that would provide as wide a margin of safety as possible.

Many scientific experts with specialist knowledge contribute to an EIA. Indeed EIAs could be used to consolidate and integrate scientific information.Even in the absence of scientific uncertainty, how does one aggregate and integrate scientific information across disciplines, government agencies and interests? Synoptic rationalists suggest Benefit-Cost Analysis, Cost Effectiveness Analysis, Multi-objective Utility Analysis etc. I shall discuss synoptic rationality below. For the present it should be noted that the hallmark of good science is replicability of experiment with identical outcomes. When the science and technology are well established and the consequences are not the subject of controversy and uncertainty, the political decision makers could make well informed decisions, confident that the scientific "competence" is not questioned. When the scientific information is not replicable in the strict sense, then the EIA should note that and the EIA process should allow the nonscientific community to be part of the interpretation of scientific information. In circumstances where risk and uncertainty are significant, the political decision-makers need to rely, to a significant extent, on judgement.The community may expect to be given the opportunity to contribute to that "best judgement". It is more reasonable to agree to prescriptive conclusions when they are not the subject of scientific controversy. When decisions are based on best judgement rather than incontrovertible scientific evidence, the EIA process should be extended to allow for more community involvement.

RATIONAL APPROACHES TO DECISION MAKING

The scientific content of EIA is the basis for a second step or component of EIA, viz. the social assessment of the consequences of the project. Ideally in this process, one needs to transform the scientific understanding of the project and its consequences into political outputs

or, as Lasswell (1958:208) puts it, "who get what, when, how". The essence of political decisions is that they have to satisfy the interests of value groups in the community. Therefore, the decisions do not only allocate natural resources among the different value groups; policy-making and political choice are, in the words of Easton (1965), "authoritative allocations of values for a society". In brief, the assessment of the social consequences of EIA alternatives is value-based or interest-based.

There have been numerous theoretical models for making rational decisions. The classical models were developed by Simon (1947) and March and Simon (1958) and will be referred to below as synoptic rationality. In order to meet some of the criticism of synpotic rationality, another set of conceptual models was developed ranging from bounded rationality and satisficing (as opposed to optimizing) (Simon 1957) to incrementalism or "muddling through" (Lindblom 1959, 1979).

Synoptic rationality assumes that the decision maker has before him the whole set of options, all the consequences of each alternative as well as a preference ranking of all consequences. The decision maker selects the alternative that leads to the most preferred consequence (i.e. maximizes the utility). Even if the decision maker has comprehensive information on options and consequences, there is the non-trivial problem of finding agreement on a single set of collective values which are to be maximized (Schwarz and Thompson 1984).

Optimization in decision making gives the expert and politician the legitimacy that is required in making public choices. Making a decision that best matches the one option to the preferred ends has a ring of authority and finality to it. Unfortunately, synoptic rationality has two sets of weaknesses that are very relevant to EIA. First, the comprehensive information set required for reaching the optimal decisions is a rarity. As Lindblom (1979) points out, "The best we can do is partial analysis"; there are too many options, too many consequences and too many uncertainties about future conditions. Second, where there is disagreement among value (or interest) groups, there is also a guarantee in advance that the decision-maker has no unique preference ranking of all consequences. Of course, synoptic rationality still has a place in decision analysis; it is an indispensible means to find the best solutions to achieve preferred ends. If there is more than one set of preferred ends, then the question is raised, "Whose preferences come first?"

Lindblom's incrementalist model of policy-making substitutes "successive limited comparisons" for the comprehensive model of synoptic rationality and its variations (e.g. bounded rationality, satisficing, and iterative, mixed scanning). This approach emphasizes "partisan mutual adjustment" in decisionmaking and puts divergent interests at centre stage.

For present purposes it is sufficient to note that whether risk assessment and EIA attempt a comprehensive or more limited inquiry, the method for selecting means and ends should reflect any conflicting interests or disagreements about values in order to maintain the credibility (i.e. the competence) of the process. The most useful output of rational decision-making techniques (e.g. multi-attribute utility analysis, benefit-cost analysis, cost-effectiveness analysis) is to demonstrate through sensitivity analysis how the outcomes are dependent on

the assigned priorities of the interest groups. In other words, the rational decision-making technique becomes not so much a tool for selecting one outcome but a tool for exploring sets of means and ends.

EQUITY IN EIA AND RA

One of the fundamental deficiencies of synoptic rationality and the incrementalist model of decision-making is that they do not require that decisions be morally justifiable. The rational model in particular is presented as being neutral with respect to distribution. Where risk and uncertainty are inherent in the project, EIA should explicitly attempt to take into account not only the "size of the pie" (i.e. efficiency) but also the distribution of benefits and risks (i.e. who pays and who benefits?). As Schulze and Kneese (1981: 81) put it:

> "Ethical questions as to justifiability of imposing risks to life and health on people, possibly people in future generations, and whether it is proper to apply conventional discounting procedures when it would effectively wipe out any consideration of the long-term future, assert themselves strongly."

In traditional benefit-cost (or risk-benefit) analysis it is enough to assume that benefits exceed costs. The costs and benefits are not weighted and the gainers do not in fact have to compensate the losers. Schulze and Kneese (1981) identify a useful set of ethical systems (Table 1) that could be used to compare the outcomes of a benefit-cost analysis that involves **uncompensated** risk. The gains are assumed to be equal to the costs but they accrue to or are borne by different sets of people. If we assume a Libertarian (or Paretian) ethic, then uncompensated risk makes someone worse off and the project would be rejected no matter what the initial distribution of wealth is. The egalitarian ethic would accept the decision only if the risk imposed on B makes the distribution of wealth and income more equal. Even the much-maligned utilitarian criterion rejects **uncompensated** risk when B is worse off than A to start with.

Table 1. Uncompensated risk A imposes on B

Ethic	$Gain_A$ =	$Cost_B$
	$Y_A > Y_B$	$Y_A < Y_B$
Traditional Benefit-cost Analysis (linear utility)	accept	accept
Utilitarian (concave utility)	reject	accept
Egalitarian	reject	accept
Elitist	accept	reject
Libertarian (Paretian)	reject	reject

Y_A, Y_B refer to initial distributions of wealth
Source: Schultze and Kneese 1981:86

The current dominant ethic in risk-benefit or cost-benefit analysis for public projects is "unweighted" (or linear) utilitarianism, e.g. majority of votes, willingness-to-pay etc. This ethic does not protect the interests or rights of a minority. While it would be difficult to argue that a minority should be allowed to block joint action on a public project, it could also be validly argued that the majority should not impose **uncompensated** risk on an objecting minority. It would also make sense to examine the distributive implications of public choices about acceptable risk decisions from different ethical perspectives for two reasons. First, it would be unfortunate if EIA were also an instrument for making the country less "ethical" (e.g. less egalitarian or less elitist, depending on one's ethical preferences). In this context the ethical-distributive implications of risk in EIA should be one more piece of information developed in the EIA for the political decision-maker. Second, the political support for joint (i.e. public) decision would be increased if compensation for risk were incorporated into decisions irrespective of the particular ethical viewpoint.

COMMUNITY COMPENSATION FOR RISK IN EIA

The argument so far points to the desirability of extending EIA so that risk be more explicitly taken into account and of explicating the equity implications of including risk in EIA. This section addresses the role of community impact mitigation in such an expanded EIA. If the criteria of scientific and technical competence and cost-effectiveness are met, how could risk decisions be made more equitable and more acceptable to the community most closely affected by the decision?

In a recent review (Grima 1985), I discussed the more traditional modes of public participation and argued that public involvement is an interconnected set of activities by four principal classes of actors, namely, Government, the Courts, Appointed Boards, and the Public. The use of the vote and of litigation are far more powerful means of involvement than participation in information-education activities and consultation activities. In the rest of this chapter I shall focus on less conventional modes of public participation, i.e. those that imply a sharing of both power and benefits in acceptable risk decisions.

BEYOND THERAPY: SHARING POWER AND BENEFITS

In a pluralistic society such as a liberal-democratic-capitalist country conflict is not only inevitable, it is also a healthy sign that competing interests are not being suppressed. In EIA and risk management, public hearings and other traditional modes of public consultation (public inquiries, mass media exchanges, open houses, advisory groups etc.) provide means for communicating data, information and priorities. Ultimately conflicts are resolved either in court through litigation or at the ballot box. These two means of last resort are too blunt and too costly to be used often; litigation takes time and strains future relationships even when the legal system already has some laws and precedents that are clearly applicable to the issue at hand.The electoral process should not be distorted by single issues (cf. abortion, nuclear disarmament) since we elect a government for several years to cope with a multitude of issues. In a real sense we vote for a leadership we trust to deal with issues that may not yet have emerged as political priorities.

The inclusion of risk analysis in EIA introduces scientific uncertainty, benefits and costs that may affect future generations as well as real differences in objective and perceived risks. These elements reinforce the argument for using conflict resolution modes that go beyond the traditional modes of public hearings and advisory groups.

Negotiation, mediation, and compensation in addition to mitigation are extensions of the current EIA participatory and consultation programs, all of which imply some degree of persuasion, cajoling, dialogue, bargaining and bluff. What is different in formal negotiation, and mediation, is that differences are resolved through voluntary bargaining by opposing parties who reach mutually acceptable terms. In essence, the parties are not merely consulted, they do not just participate (or be involved or express preferences and opinions); they share in decision-making and expect to share in the benefits of the project. This formal bargaining goes at least some way towards making the risk less involuntary and uncompensated, therefore making the risk more "reasonable" as Burton et al. (1982) put it. The formal bargaining also gives parties the opportunity to establish the "facts" (e.g. impacts, uncertainties attached to predictions, ways of expressing risk). It is this opportunity to examine the validity of sources, techniques and initial assumptions that makes the "acceptable risk" decision more reasonable (cf. Straus 1979).

It should be stressed that negotiation, mediation, consultation, litigation and other means of resolving disputes are means to an end. The bottom line consideration is community "compensation" either through impact mitigation or through other means. The most basic, significant and essential form of community compensation is impact mitigation. Other forms of compensation should not be considered **substitutes** for mitigation but simply as means to make the community derive **additional** benefits for bearing a larger share of the risk.

The main objective of EIA is to identify the environmental and social impacts of a project and to recommend amendments that would mitigate such impacts.

> "It is important to stress that the real accomplishment of NEPA is not the 8,000 EISs that have been completed, but the improvements both in the planning process and in the ensuing projects that have resulted" (Hirsch 1984:3).

Of course most or all recommended improvements in projects cost time, effort and money to the proponent; in addition these recommended changes benefit individuals and interest groups who would otherwise suffer greater losses. Therefore EIA, by definition, already includes the notion of mitigation; since this mitigation benefits some interest (or value) group other than the proponent, then this is really a form of compensation. Community impact mitigation in EIA should be made more explicit and the mode of public participation be extended to encompass community impact mitigation including compensation in monetary terms or in kind, particularly in decisions involving acceptable risk. The objective is to link the public participation that EIA already relies on, with pragmatic ways to implement "community impact mitigation".

There is an emerging and growing literature on the principles and applications of various forms of bargaining, negotiation, mediation and consensus building for resolving environmental disputes. (For a brief

history of the growth of this field see Bingham 1984.) The common characteristic of these approaches is that they all involve active and voluntary participation by the parties in the dispute to resolve differences through mutual agreement with or without the assistance of a third party. As Bingham (1984) points out, the voluntary nature of these processes sets them apart from litigation, administrative procedures and arbitration where consensus is not an objective. Of course, consensus building is not necessarily better than litigation and political actions; each approach has its place in the set of available strategies and each could be better in different circumstances.

NEGOTIATION AND MEDIATION

Negotiation is the alternative that has been demonstrably effective in labour conflict resolution and in settling international disputes, some in the environmental field, (cf. two disputes between Canada and the US: Trail Smelter dispute in the 1940s, Skagit in the 1980s).

Institutionalized negotiation would more explicitly recognize scientific and technological uncertainty and move more decisively towards a share of the benefits to those interests who would normally be required to bear uncompensated risks. In the Canadian context one should emphasize the ethical-equity aspects of voluntary dispute resolution since in Canada we have relied less on litigation to settle disputes and therefore on one hand this means that there is less incentive to negotiate in order to avoid litigation; on the other hand, since litigation is not so readily available, negotiation becomes an attractive alternative mode for achieving wide acceptance of joint (i.e. public) decisions. One could also note that litigation is best suited to resolving disputes about facts or correct procedures. Conflicts that reflect different priorities, trade-offs and values require "mutual partisan adjustment" and are, therefore, suitable candidates for negotiation or bargaining.

In environmental mediation, the bargaining takes place with the help of a third party (i.e. a mediator outside the legal court system). Why and how would environmental mediation help in creating public consensus to resolve the hazardous waste-siting problem and similar difficult-to-site hazardous facilities? That we need to create more public trust and consensus should no longer be in doubt, although it is common to regard the NIMBY syndrome as an unfortunate, unnecessary and untenable position rather than as a natural reaction of determined opponents aroused about direct impacts that threaten their economic base or their quality of life or simply incensed that their locality has to bear a disproportionate amount of the risks. EIA can help to address this issue particularly in the context of its "procedures for public participation, conflict resolution" (Hirsch 1984) and mitigation.

In environmental negotiation, mitigation of undesirable impacts is, of course, important; in addition, the parties concerned are given the opportunity to articulate and weigh trade offs. Kunreuther and Linnerooth (1983) report that economic benefits to the region and the existing rate of unemployment were significant factors in determining acceptable risk. The township of Vaughan (Ontario, Canada) withdrew its objections to a municipal landfill site for Metro Toronto in part because the proponents met most of its conditions regarding mitigation and compensation (Cameron

and Bordessa 1982). In Tacoma, Washington, the public played a useful role in the decision not to close a copper smelter which emits substantial amounts of arsenic.

Bingham (1984) reports a high success rate for negotiation-mediation (79% for site specific cases, 75% for policy dialogue). In this case, success is defined as reaching a mutually acceptable decision. The parties supported the implementation of the decision in 80% of the case-specific disputes and 44% for decision dialogues. In addition, negotiation-mediation enhances communication among the parties. (In litigation the disputants tend to stay apart.)

This process of mediation-compensation also needs time. After reviewing six cases of environmental mediation, Talbot (1983:91), summarizes the opinion of the six mediators:

> "mediation becomes feasible when a conflict matures to the point where the issues are clearly defined, the various sides perceive the existence of a balance of power, and they perceive that their objectives cannot be achieved without negotiations".

However, Talbot goes on to point out that the motivation process of mediation is enhanced when a public agency provides encouragement or applies pressure.

Other conditions that enhance the chances of successful mediation are:

1. the emphasis by the parties on legal-economic interests rather than on broad policy or philosophical questions;
2. a restricted number of competing interests and a focus on specific sites and projects (Talbot 1983).

One important feature of the voluntary resolution of environmental disputes is especially relevant to acceptable risk decisions. In risk, the "facts" themselves are often in dispute due to, say, scientific uncertainty or alternative methods of analyses or methods of presenting findings. Negotiation and mediation provide the opportunity to the parties to examine data sources, comment on the suitability of techniques and on the validity of asssumptions. This negotiation of the "facts" in the dispute involves science and scientists in what may be called a social process of "summing up" (Light and Pillemer 1984) as opposed to the conventional hypotheses testing process. A consensus has to be built not only about societal objectives but also about the scientific basis for the decision. A necessary (but not sufficient) condition for sucessful negotiation/mediation is agreement about the scientific basis for the decision, an analogue to what Fisher and Ury (1983) call the need for a "one-text procedure" for mediation and negotiation.

FUTURE DIRECTIONS

I would like to highlight two important issues by way of conclusion; these two issues are likely to be the focus of attention in arguments about the theory and the implementation of community choice in risk management and EIA.

The agency or agencies responsible for EIA play a critical role in ensuring that "acceptable risk" discussions are not only technologically sound and cost-effective but also reflect the priorities of the various interest groups. The EIA process has been successful in meeting the first two objectives but the process of public participation in EIA has been too narrow to allow voluntary resolution of disputes through negotiation and mediation. The agency or panel finds itself in the position to "decide".

Are agencies ready to broaden the public consultation process to allow environmental mediation and negotiation leading to compensation? The Canadian Environmental Assessment Research Council (CEARC) or the Federal Environmental Assessment Review Office (FEARO) or an Environmental Assessment Board, for example, could play a catalytic role by supporting skilled negotiators; disseminating the process, accomplishments and failures; and funding research into the institutionalization of negotiation as a way of resolving disputes in EIA. For example, although mediation need not be expensive, the source of funding may create problems for the independence of mediators. Should agencies such as FEARO/CEARC or U.S. EPA encourage mediation to the point of providing funds?

The second issue concerns the principle and the application of community compensation within the context of voluntary dispute resolution. Communities already have to negotiate and, if need be, litigate to obtain the highest possible level of mitigation of adverse impacts. It should be clearly understood that community compensation does not imply any relaxation of the principle that the fullest measures of mitigation possible should be implemented. Otherwise compensation could amount to an offer of a bribe to accept a lower level of mitigation and the process could deteriorate to the level of "cheque-book participation". But given that adverse effects have been reduced to the lowest possible level, there could be scope for negotiating a compensation package that would make the community in some mutually acceptable way a gainer and not just the unlucky recipient of hazards in its backyard.

The compensation need not be in the form of enhanced sport, health, cultural or educational facilities. Probably the single most important loss to the community is the loss of "security". Therefore, the compensation package should include a mutually acceptable monitoring process and the commitment to compensate any losses from the operation or malfunction of the facility for the foreseeable future.

It is unlikely that disputes about risks in EIA will just fade away; as Fisher and Ury (1983) point out, conflict is a growth industry. The gap between agencies who have to make "acceptable risk" decisions and the communities who have to bear the consequences can and should be narrowed. It would be helpful to foster a consensus-enhancing process of community involvement such as negotiation or mediation; and to ensure a mutually acceptable level of mitigation and compensation, including a guarantee of commitment and care for the foreseeable future.

A better understanding of the role of the community in decisions about risk and particularly the role of community choice could channel more energy into improving the bargaining process which results in project design changes, i.e. mitigation of impacts. EIA includes mitigation as part of its process. However, decisions about risk would be more acceptable if conflict resolution and community impact mitigation become more

explicit objectives of EIA and if the instruments for exercising trade-offs provide the community with opportunities to share in the making of decisions and in the allocation of the benefits. Community impact mitigation, including negotiation and mediation would go a long way to making risk decisions not only technologically competent and economically sound but also more equitable and more acceptable.

REFERENCES

Bingham, G. 1984. Resolving environmental disputes: a decade of experience. Executive Summary. The Conservation Foundation, Washington, D.C.

Burton, I., C.D. Fowle and R.S. McCullough (Eds.) 1982. Living with risk: environmental risk management in Canada. Institute for Environmental Studies, University of Toronto, EM-3.

Cameron, J.M. and R. Bordessa. 1982. Solid waste disposal; risk assessment or risk management? The pits at Maple, Ontario. In: Living with Risk: Environmental Risk Management, (Eds.) I Burton, C.D. Fowle and R.S. McCullough. Institute for Environmental Studies, University of Toronto, EM-3:129-143.

Easton, D. 1965. A Framework for political analysis. Prentice-Hall, Englewood Cliffs.

Fishhoff, B., S. Lichentenstein, P. Slovic, S.L. Derby and R.L. Keeney. 1981. Acceptable risk. Cambridge University Press.

Fisher, R. and W. Ury. 1983. Getting to yes: negotiating agreement without giving in. Houghton Mifflin, 1981; Penguin, 1983.

Fowle, C.D. 1986. Information needs for environmental risk assessment and management. In: Conceptual Trends and Implications for Risk Research. Report of the Task Force Meeting on Risk and Policy Analysis, (Eds.) C.T. Miller, P.L. Kleindorfer and R.E. Munn. IIASA, Austria, Publication CP-86-26, pp. 168-189.

Griffith, C., B. Bauer and A.P. Grima. 1981. Theory and practice of alternative mechanisms for environmental quality management: a bibliography. Council of Planning Librarians.

Grima, A.P. 1972. Residential water demand: alternative choices for management. Department of Geography, Research Publ. 7. University of Toronto Press, Toronto.

Grima, A.P. 1985. Participatory rites: integrating public involvement in environmental impact assessment. In: Environmental Impact Assessment: the Canadian Experience, (Eds.) J.B.R. Whitney and V.W. Maclaren. Institute for Environmental Studies, Univ. Toronto, EM-5:33-51.

Grima, A.P., P. Timmerman, C.D. Fowle and P. Byer. 1986. Risk management and EIA: research needs and opportunities. Canadian Environmental Assessment Research Council, Cat. No. En 107-3/5-1986E, 18 pp.

Hirsch, A. 1984. Effectiveness of environmental impact assessments in reducing community tensions related to difficult-to-site facilities. Paper presented at the NATO Committee on the Challenges of Modern Society Seminar on Land-Use Planning for Defense Activities, Munich, F.R.G., September 19-21, 1984, 11 pp.

Kunreuther, H.C. and J. Linnerooth. 1983. Risk analysis and decision processes: the siting of liquified energy gas facilities in four countries. Springer-Verlag, Berlin.

Lasswell, H. 1958. Politics: who gets what, when and how? Meridian Books, Cleveland.

Light, R.J. and D.B. Pillemer. 1984. Summing up: the science of reviewing research. Harvard University Press, Cambridge.

Lindblom, C.E. 1959. The science of muddling through. Public Administration Review, 19:79-99.

Lindblom, C.E. 1979. Still muddling, not yet through. Public Administration Review, 39:517-526.

Lowrance, W.W. 1976. Of acceptable risk: science and the determination of safety. William Kaufmann, Los Altos, California, 180 pp.

March, J.G. and H.E. Simon. 1958. Organizations. Wiley, New York.

Paine, R.L. and A.P. Grima. 1984. Abatement of SO_2 emissions: what are the costs? Institute for Environmental Studies, University of Toronto. Annual Report pp. 55-61.

Schulze,W.D. and A.V. Kneese. 1981. Risk in benefit-cost analysis. Risk Analysis, 1(1):81-88.

Schwarz, M. and M. Thompson. 1984. Beyond the politics of interest. International Institute for Applied Systems Analysis, Laxenburg, Austria, August, 1984. Draft Working Paper.

Simon, H.E. 1947. Administrative behaviour. Macmillan, New York, 1st Edition.

Simon, H.E. 1957. Administrative behaviour. Macmillan, New York, 2nd Edition.

Straus, D.B. 1979. Managing complexity: a new look at environmental mediation. Environmental Science and Technology, 13(6).

Talbot, A.R. 1983. Settling Things: Six Case Studies in Environmental Mediation. The Conservation Foundation, Washington, D.C.

CHAPTER 8
GIVE AND TAKE: THE ROLE OF REGULATORY DECISION-MAKERS IN RISK ANALYSES FOR ENVIRONMENTAL IMPACT ASSESSMENT

Peter N. Duinker
School of Forestry, Lakehead University, Thunder Bay, Ontario[1]

INTRODUCTION

Environmental impact assessment (EIA) can be viewed partly as a structured process for advising decision-makers of the environmental consequences of specific development proposals. Thus, it seeks to inform decision-makers of consequences they may not predict and of the perceptions that various interested parties have of those consequences. Most development proposals are so complex, affected parties so diverse, and environmental impact forecasting so technically demanding that decision-makers use processes like EIA to clarify potential consequences before making development decisions. This chapter explores the kinds of information that might be most useful to regulatory decision-makers in their evaluation of development alternatives, and also examines the regulators' role in directing and shaping an EIA. In any specific EIA, regulators and analysts can collaboratively develop and provide the regulators' information requirements on environmental risks; this "give and take" is the focus of the following discussion.

The framework is based on the assumption that the main parties that need to be considered here are regulatory decision-makers and analysts. Where appropriate, developers and public intervenors, also important parties in EIA matters, are brought into the discussion.

I start with a set of assumptions as background to the arguments developed later. These starting points clarify terms such as: decision-makers and analysts; the structure of an EIA; risks and impacts; environmental risks; and environmental risk assessment. Then, the framework is developed in terms of the classes of information that should flow between regulatory decision-makers and analysts:

(a) when an EIA gets started, to structure the problem appropriately and to give analysts proper direction for forecasting; and
(b) when the regulators require impact (risk) forecasts, to provide them with a firm basis and understanding for the evaluation tasks they must undertake in actual decision-making.

Although impact forecasting is clearly a crucial element of EIA (Duinker 1986; Duinker and Baskerville 1986), it is not the theme of this paper, and it is discussed only briefly with respect to scientific rigour. Post-decision activities in EIA, including monitoring (Duinker 1985b) are also discussed only briefly because the focus here is on advising regulators prior to making decisions.

---

[1] The paper was written while the author was with the Faculty of Forestry, University of New Brunswick, Fredericton

SOME STARTING POINTS

RISKS versus IMPACTS

In EIA, the term impact usually refers to a consequence **certain** to occur to some degree, whereas risk has been considered as a consequence that **might** occur. In this sense, both impacts and risks have a magnitude of consequence and a probability of occurrence, but impact becomes a special case of risk where the probability of occurrence is virtually one. I will treat the terms risk and impact as synonyms meaning an effect or consequence specified, at minimum, by **magnitude** and **probability**. Nevertheless, I recognize that risks with very low probabilities of occurrence and serious consequences present major difficulties in perception to all parties concerned. Many such low-probability risks have never been experienced before and, therefore, we have some trouble thinking clearly and comparatively about very low probability events (Dooley and Burton 1983; Whyte 1983).

Predicted impacts and risks of development can be defined with a minimum of two forecasts: one with a specific intervention (development alternative) proposed, and one without. Impacts of that specific intervention are then calculated as the difference between these forecasts.

DECISION-MAKERS AND ANALYSTS

In this discussion, the term decision-maker in EIA includes:

(a) senior officials within agencies, companies, or institutions proposing development (i.e. developers); and
(b) senior officials and advisors within governments that control development (i.e. regulators).

Analysts are scientific experts retained by developers or regulators to undertake and review analyses, especially impact forecasts. The term "intervenors" represents a large and diverse group, including private and government individuals and groups who are often invited to comment on EIAs and their review.

Let us look at the main role and objectives of each actor in turn. The developer actually proposes, initiates and implements development. She/he usually operates in an opportunistic mode, pursuing objectives and seeking gain while recognizing constraints of many kinds. The regulator is in a position to permit and control development in specific respects. She/he usually operates in a prevention mode, developing and applying constraints on development so as to protect environmental values within her/his mandate. Analysts, retained by the developer, the regulator or the intervenors, are scientists who are expected to produce defensible analyses including the technical information desired by their client. Finally, intervenors become involved by providing opinions on:

(a) terms of reference for a public EIA review;
(b) terms of reference for an EIA (or guidelines);
(c) the development proposal itself; and
(e) the acceptability of the predicted risks.

This chapter concerns knowledge needs of the regulators. However, because there are two main groups of decision-makers in the customary formal EIA (i.e. the regulator and the developer), one cannot speak of the regulators' information needs without referring to the interactions among the other three actor groups (i.e. developer, analysts and intervenors) themselves in addition to the interactions between regulators and those three groups. This follows from the fact that while the regulator can require an EIA of the developer and can control the public review, the developer is charged with conducting the studies and producing the EIA. Thus, both groups of decision-makers control the nature of the information required of analysts.

THE STRUCTURE OF AN EIA

Let us consider EIA as a process consisting of four phases: (a) scoping; (b) forecasting; (c) evaluation; and (d) follow-up. This matches closely a scheme for risk assessment put forward by Somers and Krewski (1983), where the phases are called: (a) risk identification; (b) risk estimation; (c) risk evaluation; and (d) risk management.

Scoping involves activities such as identifying the main actors in the EIA, the development design and its range of alternatives, and the environmental systems to be studied. This phase allows the developer to explain the proposed development to the other parties, and gives intervenors a chance to comment on the direction the EIA should take. It also permits the regulator and developer to provide analysts with a definitive statement of the information required from a risk assessment. The **forecasting** phase is the purview of analysts who, as scientists, undertake the scientific investigations necessary to provide the forecast information specified in the first stage. Once the analysts have communicated the findings of forecasting studies to the developer, regulator and intervenors, the parties engage in **evaluation** where they, among other things, argue the relative importance or significance of the predicted impacts including risks. Finally, once the development is approved and initiated, the **follow-up** stage includes measurement of specific environmental phenomena to reduce uncertainty in the forecasts, monitoring and enforcement of terms and conditions set in development approval, and appraisal of the successes and failures of the EIA as a process for advising decision-makers.

ENVIRONMENTAL RISKS

In the early 1970s, the word "environment" in EIA meant natural or biophysical environment including wild flora, fauna and ecosystems. Of course, we have concern for the natural environment mostly for cultural, social and economic reasons (Duinker and Beanlands 1986). However, as the decade of the 1970s progressed, legislation began to include social and economic environments as well in EIA. Assessments then included direct relationships between proposed developments and social/economic impacts such as employment, community social structure, community services and infrastructure. In this chapter I use the term "environment" to apply only to the natural or biophysical environment.

THE FRAMEWORK, PART I - SCOPING AN EIA

The first phase of an EIA, as defined above, is essentially a problem-formulation phase which needs careful attention if analysts are to be given meaningful direction for their time-consuming and expensive studies of the biophysical environment. The following sections are structured around a series of questions that analysts should pose before engaging in their scientific studies. I propose that it is the joint responsibility of the regulator and the developer to provide answers to these questions. In the interest of fair play and expeditious later review, the regulator and developer make reasonable efforts to secure the opinions of intervenors on all the questions posed below, **before** terms of reference for an EIA are formally established; the federal EIA process in Canada includes such a provision. Here, then, are some directions the decision-makers need to provide for an EIA.

WHAT IS THE DEVELOPMENT?

Analysts in EIA are faced with systems-modelling problems where the objective is to search for and specify cause-effect relationships between a proposed development and specific aspects of the environment. Obviously, analysts need a clear, explicit and specific understanding of the nature of the development, for it **must** be the first component or set of components in forecasting models if it is understood to be a "cause" of impact.

There has been little agreement among analysts as to the level of detail required about a proposed development (Beanlands and Duinker 1983). Many analysts have tended to ask for complete "nuts and bolts" descriptions of developments, not unlike their tendency to describe natural environments in every intimate detail. Beanlands and Duinker (1983) advocated looking at a development from an "ecological" point of view, that is, trying to determine what key factors in a development's structure and implementation really count with respect to the responses in the target environmental system. While the parties work out other matters in this start-up phase of an EIA, analysts will undoubtedly discover that additional details on a proposed development are needed as they search for cause-effect relationships. Thus, the specification of the development being assessed really should be an ongoing process consisting mainly of questions from the analysts and answers from the decision-makers, particularly the developer proposing the project.

WHAT ARE THE DECISION VARIABLES?

EIA not only helps decision-makers make informed decisions on **whether** to allow or proceed with development. Many of the development decisions which need an EIA deal with the conditions that are attached with the approval to proceed. These conditions demonstrate that developments are rarely unalterable packages of only one configuration, but often are assemblages of structures and plans, many parts of which can be altered while still meeting the objectives of the project. Let us call those parts decision variables, since they are development variables whose values the decision-makers can set as they choose. So-called mitigation measures in EIA are really decision variables - they are ways of altering developments so as to alter environmental risks.

While specification of the decision variables is the responsibility of the developer, occasionally the regulator can establish the identity and nature of decision variables, e.g. when insisting on specific mitigation measures

designed to prevent deterioration of some environmental asset. Examples include minimum flow requirements for protection of fish in impounded rivers, uncut forest strips adjacent to streams, and wildlife passes under highways. In such cases, the regulator has not bound the developer directly with protection of the specific environmental asset, but rather has required the developer to adopt an action or set of actions to provide the protection desired.

For some kinds of developments, the range of decision variables may be inherently large. For example, a plan to harvest timber for several decades from a block of Crown land involves many decision variables, including how to cut, where to cut, when to cut, how much to cut in any one time period, etc. On the other hand, some kinds of developments are almost cast in stone even on the drawing board. For example in the proposal for a second nuclear reactor at Point Lepreau in New Brunswick, features such as the location and type of reactor, and the support systems (e.g. cooling water systems) are either approved as proposed, or not at all.

Since analysts attempt to build forecasting models that can be used to generate information useful to decision-makers, (i.e. models that forecast what might happen in the environment if certain specific decisions are implemented), they need to know from the decision-makers what the decision variables are, and within what ranges they are feasible in the development being assessed. The analysts must explicitly model the decision variables in precisely the way the decision-makers have control over them.

The identification of decision variables, their feasible ranges and functional connections with specific components of the natural environment are seldom simple matters; insightful collaboration among the actors is required to discover jointly what the practical decision possibilities are. Ultimately, though, it must be the decision-makers (regulator or developer, or both as appropriate) who instruct the analysts as to the decision variables and their ranges for which forecasts must be made; the intervenors need to have a persuasive argument to sway this type of decision. This is discussed further in Part II of the framework.

WHAT ARE THE ENVIRONMENTAL PERFORMANCE INDICATORS?

The early EIAs typically attempted to be comprehensive about the environment (and thus turned out to be inconclusive about everything); it is generally agreed now that an EIA needs to focus strongly on a relatively small set of components of the biophysical environment. Let us examine the specification of these components in two parts. First, the components need to be identified. I call these components **environmental performance indicators**, because they enable the decision-makers to judge the relative performance of each development configuration in relation to the environment. Secondly, for each environmental performance indicator, one or more **measures** need to be chosen as the term or terms in which forecasts for the environmental performance indicator will be stated. For example, if deer are identified as an environmental performance indicator in the assessment of a proposal for silvicultural use of herbicides, we could express the performance of the deer in terms of: (a) total population numbers; (b) deer density; (c) rate of mortality, or number of deer deaths; (d) quality of deer flesh; (e) size of trophy deer; etc. (Duinker 1986).

Similarly, for risk to human health, there are several combinations of probability and magnitude of potential risks for each component of a potential-

ly affected population. The terms chosen by decision-makers to express forecasts for any environmental performance indicator are overriding determinants of how analysts construct their causal models. Thus, the specification of these measures is an essential precursor to a forecasting exercise.

The list of performance indicators that a regulator is interested in will undoubtedly differ from a developer's. This is to be expected, considering the different mandates, objectives and constraints of these decision-makers. The developer is interested primarily in a successful and technically feasible economic proposition, while the regulator, in the case of the biophysical environment, is interested in the protection of certain environmental assets. For an EIA to be a useful source of information to both regulators and developers, both sets of environmental performance indicators need to be included. However, one must avoid huge arrays of indicators and measures, and instead focus on determining those to be examined in detail.

Again, the identification of environmental performance indicators and their measures should not take place in a one-way, once-only communication from decision-makers to analysts. Frequent consultation may be necessary because the decision-makers may not know all the possibilities at the outset, and analysts may initially have unrealistic notions of the decision-makers' needs. But it must be the decision-makers who ultimately decide on environmental performance indicators and their measures - after all, results from a risk-forecasting exercise are primarily for the purpose of informing the decision-makers. It is here, though, where intervenors can make important inputs into an EIA. Intervenors are often asked questions such as "what issues and concerns should be dealt with in this EIA?", and are being increasingly asked to participate in Canadian EIA, especially at the national level. However, these "issues and concerns" need to be translated into specific and measurable environmental performance indicators before they can be adopted by either the decision-makers or the analysts.

Analysts have an important role in identifying environmental performance indicators and measures. As scientists, the analysts may have special insights into specific possible performance indicators and measures that may be overlooked by a lay public, and these should be explained to the decision-makers. As well, analysts are in a position to advise the decision-makers whether there may be any insurmountable difficulties in preparing credible risk forecasts for specific performance indicators or measures that the decision-makers may desire.

To conclude, the explicit identification of environmental performance indicators and their quantitative measures is absolutely essential and should take place in collaboration among all parties concerned, with plenty of opportunity for feedback, discussion and alteration.

WHAT BOUNDS IN TIME AND SPACE?

Any analysis of the effects of a proposed development applies within temporal and spatial limits which should be explicitly specified. There are two kinds of such limits. One kind we usually call boundaries, meaning the "outer" limits of an analysis. Temporally, this refers to how far into the future we make our projections. Spatially, it refers to how far afield we go in looking for impacts. Anything temporally or spatially beyond these limits is explicitly left out of the search for impact. The other kind of limit is an "inner" limit, referring to the temporal and spatial resolution of an analysis.

What levels of temporal and spatial detail are fine enough for the analysis? Anything finer is explicitly not dealt with, and is assigned the average condition of the smallest temporal or spatial unit (i.e. the variation within such units is ignored).

As a simple example, we could examine a proposal for silvicultural use of herbicides on a large forest. The spatial boundary could be the limits of the forest under consideration (everything outside is ignored), and the spatial resolution could be the level of forest stand (variation within stands is ignored). The temporal boundary could be 50 years, a plausible rotation age for managed stands in eastern Canada (everything beyond is ignored), and the temporal resolution could be annual (variation within years is ignored).

With respect to outer bounds, Beanlands and Duinker (1983) advised decision-makers and analysts to pursue bounding first on an administrative basis (time and space corresponding to the decision-makers' jurisdiction), next on a project basis (time and space dimensions of a proposed development), and subsequently on a biophysical basis (time and space scales over which the affected natural systems operate).

The level of detail as well as the outer limits desired by interested parties may vary, but decision-makers must ultimately determine the explicit bounds that will apply to the production of forecast information useful in development decision-making. Analysts can certainly use finer levels of detail in generating impact forecasts, provided these can defensibly be aggregated if a coarser level is required by decision-makers. But levels of detail coarser than the decision-makers' requirements cannot serve as a basis for risk forecasting, because these can not be defensibly subdivided after analysis is complete.

WHAT LEVELS OF THE PERFORMANCE INDICATORS ARE SIGNIFICANT?

For many kinds of environmental variables, there are standards, guidelines, or targets which serve as reference points in attempting to set the levels of the variables as they are affected by development. We have many examples from fields such as air quality, water quality and radiation. For other kinds of environmental variables, decision-makers in EIA may have specific objectives for the levels of the variables. Examples might be sustainable harvest of a deer population, preservation of a scenic view, and merchantable wood harvest from a forest. For still other kinds of environmental variables, analysts may have insights into the factors controlling any such variable such that they can identify ranges within which the functional relationship is "stable", and ranges where it is unstable and prone to moving to zero or unprecendented high levels. An example might be populations of endangered wildlife species.

Any environmental performance indicator (and its measure), chosen by either the regulator or the developer, may fall into one of the three categories above. If so, it means that some specific levels or ranges in the performance indicators are special in some sense (e.g. regulated maxima or minima, objectives to be sought, or levels to avoid), and analysts should pay special attention to these levels or ranges in their risk analyses. Two important reasons for this include:

1. analysts may be in a position to investigate, design and recommend mitigation measures; thus, they would need to know the appropriate performance level of the indicator to design for; and

2. a decision-maker's tolerance for uncertainty in a forecast may be small when the performance indicator's level is forecast to be near a critical level, while it may be relaxed somewhat when the level is forecast to be orders of magnitude safely away from a critical level; for example, a forecast of an ambient pollutant concentration of 10,000 times less than the regulated level may not need to be as accurate or precise as a forecast of 10 times less.

WHAT LEVEL OF UNCERTAINTY IN THE FORECASTS?

Decision-makers know that no forecast can make the future certain. All decisions based on forecasts in EIA carry some degree of uncertainty. However, decision-makers also know that specific small elements of uncertainty about the future of any environmental performance indicator may, through insightful analysis and research, be isolated and made less uncertain; the nature of many remaining elements of the uncertainty can at least be characterized. Decision-makers can often specify in general terms the types of error they could live with in the forecasts for specific performance indicators and analysts can often respond as to whether such specifications can be met at all, and if so, at what cost in time and money. Thus, before analysts can begin a scientifically defensible and practically useful risk forecasting exercise, they need to know from the decision-makers in approximate terms the nature of the uncertainty that could be tolerated in the forecasts.

HOW SHOULD FORECASTING RESULTS BE COMMUNICATED?

What makes reporting the substance of an EIA difficult is that the results need to be meaningful to and understood by a wide array of people who have an equally wide scope of information needs and comprehension of the subject matter. Again, because the EIA is undertaken primarily for informing decision-makers, they should establish the form and means in which results of risk forecasting are to be reported. There are numerous possibilities here, including many clever ways of representing messages textually and graphically. Analysts should pay special attention to appropriate and attractive means of communication for the kinds of information that will be forthcoming and for the people who will use it.

RISK FORECASTING

As mentioned previously, the way in which risk forecasting is done, while central to an EIA, is not the substance of this paper, and therefore, is discussed only briefly here (for some discussion of this topic see Duinker 1985a; 1986; Duinker and Baskerville 1986).

In the first phase of an EIA described above, there should have been considerable discussion among all parties on specific directions for the scientific studies. The information flow has been predominantly in the form of questions from analysts, suggestions from intervenors, and answers from decision-makers. The analysts can now earnestly begin risk forecasting, a set of tasks essentially scientific in nature. There may still need to be

considerable contact with the decision-makers, in case the analysts need more direction, or in case the decision-makers want to see interim results. But, essentially, the analysts alone will be working on the problem of producing scientifically well-grounded hypotheses about how specific aspects of the development under consideration may affect specific components of the environment.

The decision-makers expect the analysts to meet the specifications agreed upon before forecasting got started, or else produce convincing arguments why they can not meet the specifications. Otherwise, the decision-makers should give analysts free rein to practice their sciences in producing risk forecasts. Cautious decision-makers usually get other analysts to check the work of the primary analysts, as a sort of quality assurance or peer review. Quality science in EIA, as in other decision-making processes (e.g. Efron 1984), can only be obtained with credible scientific peer review. This leads to a very important feature that should characterize the scientific work in all EIAs - for the work to qualify as rigorous science, all essential steps must be made explicit so that other scientists can check the work and reproduce the results! Implicit "science", where analysts produce their risk forecasts **directly** from professional judgement with no formal basis laid out, is not rigorous science, if it is science at all. This does not mean that scientific forecasting is devoid of all value judgements - it means that the judgements are explicit, and that they are "scientific" judgements rather than "social" and "moral" judgements.

THE FRAMEWORK, PART II - AFTER FORECASTING, BEFORE DECISIONS

Decision-makers have asked for risk (impact) forecasts to serve as a basis for comparing development options in terms of environmental performance. When results are to be described by the analysts, the decision-makers should see, in terms of their choice, forecasts (within prescribed temporal and spatial limits and resolution) of the levels of the environmental performance indicators (in terms of the prescribed measures) over reasonable ranges of the decision variables that affect them. In addition, there are other important messages that should also be conveyed to the regulator.

Any biophysical risk can be analysed (indeed, should be analysed) in terms of a linked set of causal relationships, with the first component being an element of the proposed development, and the last one the selected environmental performance indicator. Such cause-effect sequences can range from the very simple to the very complex, depending on many factors. Each individual condition or event is either controllable to some degree (e.g. construction of a dam, accidental oil-well blowout) or is not controllable (e.g. earthquake, severe weather). As well, each condition or event can be characterized by its own probability of occurrence, and magnitude or severity. In arriving at a set of defensible risk forecasts for an environmental performance indicator, analysts will have had to relate all these conditions and events, complete with their respective probabilities of occurrence and magnitude, to each other in an explicit fashion. This done, analysts and decision-makers can move forward into discussions of where and how risks might be controlled, and how scientific uncertainty might affect decisions on controlling risks.

WHERE AND HOW TO CONTROL RISKS

Decision-makers are seldom fully aware of all the possibilities for altering a development, or altering related conditions and events, to improve

the response of environmental performance indicators. For that matter, analysts may not be fully aware of the possibilities either. Together, though, and with the intervenors, they can productively explore these possibilities, but **only if** the analysts can show the cause-effect linkages in quantitative terms, between a development and a performance indicator. The discussions can cover:

(a) where control is even possible;
(b) where effectiveness of control is reasonably sure;
(c) where control resources would be most efficiently applied; and
(d) whether control should be aimed at changing probability of occurrence or magnitude of an event to a critical level.

Dooley and Burton (1983) discussed some of these in detail. Let us examine here items (b) and (d) above.

Impact control that is robust to error and uncertainty

First, conditions or events over which decision-makers have control were called decision variables above. Such variables will have been incorporated into the forecasting analyses if they were identified early on as important determinants (i.e. conditions or events) of the future of an environmental performance indicator. The regulator should be shown the forecast response of the performance indicator for a feasible range of levels of the decision variables. This is very simply done in a two-dimensional graph for one decision variable, or for two decision variables simultaneously on a three-dimensional response surface. If the same two decision variables affect several performance indicators, then several response surfaces, each showing the response of a different performance indicator, can be displayed side by side to facilitate a very instructive trade-off analysis (e.g. Peterman 1975). Two decision variables often used in response surfaces for human health risk analysis are:

(a) duration of exposure to a toxic agent; and
(b) amount or concentration of toxic agent (Krewski et al. 1982).

Those familiar with the features of response surfaces will know the importance of plains and cliffs. A plain on the surface indicates sluggish response of an environmental performance indicator to the decision variables, such that large changes in the decision variables are needed to produce relatively small changes in the performance indicator. Decision-makers often consider these as "safe" zones from the perspective that errors in the forecasting exercise or in implementing desired levels of the decision variables are likely to have relatively small effects on the level of the performance indicator. In such cases, actions taken to reduce undesirable effects or increase desirable ones can be said to be robust to error. On the other hand, cliffs indicate a very sensitive response of an environmental performance indicator to the decision variables, such that small changes in the decision variables can produce relatively large changes in the performance indicator. Decision-makers often consider these as "danger" zones from the perspective that small errors in forecasting or in decision implementation can have drastic effects on the level of the performance indicator. In such cases, mitigative actions need to be exceptionally carefully applied for the desired effects to be achieved consistently. (See Holling (1978) and Duinker (1986) for examples of such analyses.)

Reduce probability or consequence?

Let us consider the risk picture for an environmental performance indicator, from some aspect of a proposed development, consisting of a contingency table showing a range of probabilities of occurrence and corresponding magnitudes of consequence. Risk reduction often has been focussed on reducing higher probability-lower magnitude risks (i.e. reducing system variability) with concomitant but ignored or unforseen increases in the probability of larger magnitude risks (Clark 1979). In many cases (e.g. flood control, forest fire control), this has led to a worsening of the overall risk picture over time than before control measures were instituted! For any proposal to alter risk, one must be careful to examine the risk picture comprehensively to determine whether the proposal increases risk in one part of the picture while attempting to decrease it elsewhere.

In summary, analysts need to present to the decision-makers their conceptions of the cause-effect nature of a risk, so together they can meaningfully explore ways of adjusting the risk to more satisfactory levels or distributions. Of great importance here are explorations of the uncertainty of impact controls, and of the probability-magnitude matrix of the potential impacts. As I note below, the perception of risks by different actors (e.g. intervenors) may complicate the process.

SCIENCE AND UNCERTAINTY

Biophysical impact forecasting in EIA is mainly an exercise in reducing uncertainties. These uncertainties can be categorized as:

(a) random exogenous variables, for which probability distributions can often be constructed;
(b) errors in the way relationships are structured and parameters quantified; and
(c) fundamental errors in problem and system representation (Walters and Hilborn 1978).

Since such uncertainties influence the nature of the risks that decision-makers have to deal with, they will want to know how such errors could affect the responses of the environmental performance indicators. Analysts, therefore, need first to identify and localize the most uncertain elements or components of their forecasting models. Model building will have involved structuring and linking a set of biophysical relationships. Each of these relationships should be examined for the uncertainty in the basis of its formulation, both in its general form and in the parameters that make it case-specific. As well, uncertainty in the way in which the sub-systems in the models are linked should be examined. Through sensitivity analysis with the forecasting models, the analysts would then check the response of each environmental performance indicator to systematic adjustments in the uncertain components. If the response were sensitive (i.e. small change in the uncertain component resulted in large change in response of the performance indicator), then the uncertainty would be flagged as important. If the response were insensitive (i.e. the level of performance indicator changed very little with wide adjustments in an uncertain component) then the uncertainty would be flagged as unimportant in this environmental problem-solving exercise.

Analysts will want to investigate and resolve as many of the important uncertainties as possible before communicating risk forecasts to the decision-

makers. This is done, of course, through field or laboratory research. However, analysts cannot render the future certain, and the results of forecasting will remain uncertain. The best analysts can, and should, do is to disclose to the decision-makers what the major uncertainties remaining are, as well as showing the response of the performance indicators to systematic adjustments in the uncertainties.

If a performance indicator is very sensitive to specific unresolved uncertainties, the regulator has several kinds of options open. First, a decision could be taken despite the uncertainty, erring on the side of safety in the decision variables. This could be done by identifying the worst-case scenario for the uncertain system components and setting the levels of the appropriate decision variables to obtain the desirable levels of the performance indicators. This happens routinely, for example, in the requirement for extensive add-on mitigation measures in the nuclear power industry, and in contingency plans for accidents in offshore oil production.

Secondly, the regulator could postpone making a decision to permit development until specific key uncertainties are resolved. This happened in relation to the Alaska Highway Gas Pipeline (Alaska Highway Gas Pipeline Project Environmental Assessment Panel 1979). The Environmental Assessment Panel postponed a recommendation to the federal Minister of the Environment on approval of the project until the proponent had rectified information deficiencies related to several outstanding issues such as the effects of various pipeline mode alternatives, route alternatives, construction alternatives, and impact mitigation options. Finally, the regulator could permit development as proposed and require careful monitoring of the uncertain relationships, taking further action later if monitoring suggests that undesirable effects are occurring.

EVALUATION AND PERCEPTION OF RISKS

Evaluation of risks is a critical element of EIA. Indeed, all decisions are preceded by some form of implicit or explicit evaluation exercise. In EIA, it basically entails a desirability comparison, in terms of the environmental performance indicators, of the futures of each development option being assessed, including the no-development option. It may be a rather simple process, where regulators and developers gather opinions from intervenors on the significance or importance of the predicted outcomes and then informally evaluate performance and choose one of the development options. This is characteristic of much of EIA in Canada today. Or the process may be rather complex, involving the use of quantitative decision-analysis techniques. Many of these have been advocated for use in EIA (e.g. McAllister 1980), but they have seen more application in land use and water resources planning in North America than in EIA.

As stated above, the purpose of EIA is to apprise decision-makers of the environmental consequences of proposed development. We can view the advisory information as consisting basically of two sets - a set of technical, usually quantitative information on risks as forecast by analysts and a set of information on the perception, feelings and opinions of risk from the interested or affected parties called intervenors. EIA was originally adopted through the insistence of people with concerns for their own well-being as development affected them, and for the well-being of natural flora and fauna and ecosystems as they were affected by development. Thus it is appropriate that the regulator, in making decisions on behalf of the public, canvass broadly for opinions

on the matters about which a decision must be taken. Discussions above have concentrated on the technical information set that the regulator might need. Timmerman's paper in this volume deals in some detail with the notions of perception and conception of risk, and other papers elsewhere (e.g. Kasperson and Kasperson 1983; Thompson 1983; Whyte 1983) have reported on this topic. I conclude that both sets of information are essential to the regulator. Evaluation in EIA is likely to be a trivial and confusing exercise when it cannot rely on a reasonably secure basis of quantitative technical risk information, and it is likely to be irrelevant when it does not take into account the views and opinions of the various intervenors and other stakeholders.

FOLLOW-UP

EIA practitioners are unanimous in their call for EIA to continue into the post-decision stage of surveillance and monitoring (Beanlands and Duinker 1983). Elsewhere (Duinker 1985b) I have argued the need for monitoring of environmental performance indicators for several reasons:

(a) to monitor forecasts and thus improve predictive capability;
(b) to check the effectiveness of mitigation measures;
(c) to provide early warning of undesirable change so that corrective action can be taken; and
(d) to provide evidence to refute or support claims for compensation.

Let us develop an argument from the regulator's point of view. Regulators can never know whether the **actual** performance of the environmental performance indicators under the influence of the development is consistent with the **forecast** performance unless someone measures and reports that performance. Learning will not take place, and future decisions of a similar nature cannot be provided with a firmer basis, without such measurement. The regulator needs monitoring information to determine whether critical levels (e.g. regulated maxima or minima for pollutants) are being exceeded, and thus whether action is warranted. Of course, the same argument can be followed for the developer, although the specific performance indicators of interest to the developer will likely be different from those of the regulator. If either decision-maker expects to engage in **adaptive** management of the environment, where future management decisions are responsive to and conditioned by the response of environmental indicators to implementation of previous decisions (Holling 1978; Baskerville 1985; Walters 1986), monitoring of the indicators is an essential component of the process. Adaptation is prevented by ignorance.

As with pre-decision information, the regulator will want not only the technical information on environmental performance as provided by monitoring, but also will want to gauge the evolving opinions and perceptions of affected parties on environmental risks associated with the development. This information is a necessary complement to the technical information in providing guidance to the regulator on what further actions might be appropriate with respect to the development at hand, and on prudent approaches to future similar decisions.

CONCLUSIONS

This paper has sketched the kinds of information that should flow between decision-makers and analysts in a quantitative risk analysis in EIA. Risk-related decision-making takes place in complex circumstances and decision-makers need clear and concise information from quantitative risk analysis. Briefly, at the beginning of an EIA, the regulator (and developer and intervenors) needs to give substantial direction to analysts so that results useful in the decision problem can be generated. However, analysts need to take active part in problem specification by asking appropriate questions of the regulator. Interaction is needed here because at the outset neither party is in a good position alone to structure the problem adequately. The actual production of forecasts is the analysts' task, and the regulator's interests are usually limited to ensuring that the work:

(a) meets the specifications agreed upon; and
(b) is rigorous science and submitted to peer review.

Subsequently, analysts report the forecasting results to the decision-makers, according to the specifications agreed upon, and paying particular attention to areas where environmental performance indicators are sensitive to decision variables and unresolved uncertainties. Finally, following an evaluation of alternatives, a permitting decision, and implementation of the development, analysts measure specific environmental phenomena to provide regulators with information needed for future actions.

In conclusion, I believe that regulators (and developers) will get far more useful information out of quantitative risk analysis in EIA when they:

(a) spend more time and effort giving meaningful directions to risk-forecasting studies;
(b) are served by analysts who are capable of engaging in scientifically defensible risk forecasting;
(c) insist on information-rich results from analysts; and
(d) institute an atmosphere of cooperation and open discussion among all parties throughout the proceedings of an EIA.

ACKNOWLEDGEMENTS

I thank Gordon Baskerville and Murray Paterson for commenting on early drafts.

REFERENCES

Alaska Highway Gas Pipeline Project Environmental Assessment Panel. 1979. Report to the Minister of the Environment on Alaska Highway Gas Pipeline Project, Yukon Hearings. FEARO, Hull, P.Q.

Baskerville, G.L. 1985. Adaptive management: wood availability and habitat availability. Forestry Chronicle, 61:171-175.

Beanlands, G.E. and P.N. Duinker. 1983. An ecological framework for environmental impact assessment in Canada. Inst. Resource and Environ. Studies, Dalhousie Univ., Halifax, N.S. and FEARO, Hull, P.Q. 132 pp.

Clark, W.C. 1979. Witches, floods and wonder drugs: historical perspectives on risk management. In: Societal Risk Assessment: How Safe is Safe Enough? (Eds.) R.C. Schwing and W.A. Alberts. Plenum, N.Y., pp. 287-313.

Dooley, J. and I. Burton. 1983. Risk assessment for the transport of hazardous materials. In: Proc. Symp. Assessment and Perception of Risk to Human Health in Canada, (Eds.) J.T. Rogers and D.V. Bates. RSC, Ottawa, pp.81-87.

Duinker, P.N. 1985a. Biological impact forecasting in environmental assessment. In: Environmental Impact Assessment: The Canadian Experience, (Eds.) J.B.R. Whitney and V.M. Maclaren, Institute for Environmental Studies, University of Toronto, EM-5:115-127.

Duinker, P.N. 1985b. Effects monitoring in environmental impact assessment in Canada. In: New Directions in Environmental Impact Assessment, (Eds.) V.M. Maclaren and J.B.R. Whitney. Methuen, Toronto, pp. 117-143.

Duinker, P.N. 1986. A systematic approach for forecasting in environmental impact assessment: a deer-habitat case study. Unpublished Ph.D. thesis, Faculty of Forestry, University of New Brunswick, Fredericton.

Duinker, P.N. and G.E. Beanlands. 1986. The significance of environmental impacts: an exploration of the concept. Environ. Management, 10:1-10.

Duinker, P.N. and G.L. Baskerville. 1986. A systematic approach to forecasting in environmental impact assessment. J. Environ. Management, 23:271-290.

Efron, E. 1984. The apocalyptics: cancer and the big lie. Simon and Schuster, New York, 590 pp.

Holling, C.S. (Ed.). 1978. Adaptive environmental assessment and management. John Wiley, Chichester, 377 pp.

Kasperson, R.E. and J.X. Kasperson. 1983. Determining the acceptability of risk: ethical and policy issues. In: Proc. Symp. Assessment and Perception of Risk to Human Health in Canada, (J.T. Rogers and D.V. Bates (eds.). RSC, Ottawa, pp. 135-153.

Krewsky, D., D. Clayson and R.S. McCullough. 1982. Identification and measurement of risk. In: Living with Risk: Environmental Risk Management in Canada, (Eds.) I. Burton, C.D. Fowle and R.S. McCullough, Institute for Environmental Studies, University of Toronto, EM-3:7-23.

McAllister, D.M. 1980. Evaluation in environmental planning. M.I.T. Press, Cambridge, 308 pp.

Peterman, R. 1975. New techniques for policy evaluation in ecological systems: methodology for a case study of Pacific salmon fisheries. J. Fish. Res. Board Can., 32:2179-2188.

Somers, E. and D. Krewski. 1983. Risks from environmental chemicals. In: Proc. Symp. Assessment and Perception of Risk to Human Health in Canada, (Eds.) J.T. Rogers and D.V. Bates. RSC, Ottawa, pp. 43-50.

Thompson, A.R. 1983. Science and beyond in environmental impact assessment. In: Final Report, Intern. Workshop on Environ. Planning for Large-scale Development Projects, (Eds.) J.D. Wiebe, E.H. Kuston and S. Hum. Environ. Can., Vancouver, pp. 4-18.

Walters, C.J. 1986. Adaptive management of renewable resources. MacMillan, New York, 374 pp.

Walters, C.J. and R. Hilborn. 1978. Ecological optimization and adaptive management. Ann. Rev. Ecol. Syst., 9:157-188.

Whyte, A.V. 1983. Probabilities, consequences and values in the perception of risk. In: Proc. Symp. Assessment and Perception of Risk to Human Health in Canada, (Eds.) J.T. Rogers and D.V. Bates. RSC, Ottawa, pp. 121-130.

CHAPTER 9
MANAGING UNCERTAINTY IN ENVIRONMENTAL ASSESSMENT: A PROJECT PROPONENT PERSPECTIVE

R.J. Malvern and W.M. Paterson
Environmental Studies and Assessments Department, Ontario Hydro

INTRODUCTION

The environmental assessment (EA) process in Canada, and elsewhere, has become recognized as a convenient forum to debate and resolve the myriad of trade-offs involved in fulfilling both the long- and short-term societal goals that relate to a particular development proposal. EA was originally conceptualized in the late 1960s and early 1970s to deal only with the natural environment. Increasing demands placed on the process have considerably broadened its scope. Project need and justification, technical and economic feasibility, social/community implications, socio-political acceptability, as well as some determination of the project's role in planning and development, are now understood to be integral and important parts of a comprehensive EA. The onus of acquiring and providing relevant information to support this expanded assessment of projects lies primarily with the proponent, at least initially. A logical progression in this move to a more multi-disciplinary and integrated approach to EA has been a growing concern and preoccupation with risk and uncertainty.

In recent years, risk assessment (RA) has been utilized with increasing frequency to aid in decisions ranging from very broad technological choices to ones relating to the reliability of very specific systems or even system components (Inhaber 1978; Pate-Cornell 1984). These RA applications are relevant to the environmental assessment (EA) of large-scale development projects or programs. Where considerable uncertainty exists in a project assessment, it needs to be resolved before the project is implemented or the project may not live up to its expected standards of performance.

In planning and implementing a large development project, risks and uncertainties are evident at two general levels of decision-making (Fig. 1). For the purpose of this discussion, these two types of decision-making will be termed "macro" and "micro" levels.

Macro-level decisions generally pertain to broad, societal tradeoffs or choices affecting the need and justification for a project. Uncertainty in this case relates to broad policy issues such as jobs or dollars _versus_ environmental protection, quality of life, risk/benefits of technological alternatives, pricing/forecasting, resource use/commitment, and incentive or regulatory initiatives. Most of these front-end uncertainties are difficult to handle and tend to be beyond the control of the project proponent, but they frequently have a significant bearing on the acceptability of the proposal. In essence, macro-level decisions set the stage for specific projects. When not settled in an acceptable and consistent manner at the beginning of project planning, these types of uncertainties can plague a project throughout the EA process, and can significantly delay its implementation (Malvern 1983).

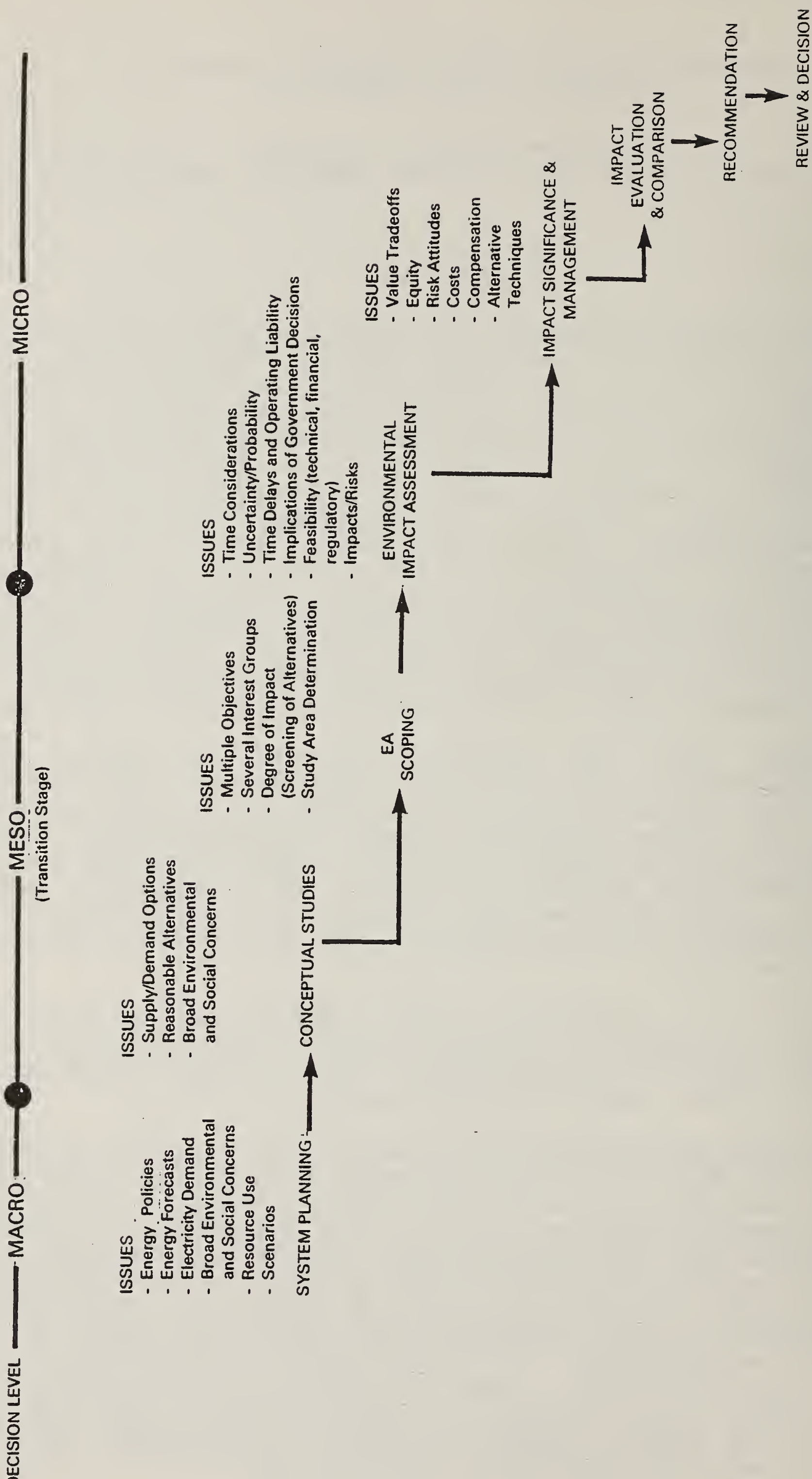

Fig. 1. An EA decision-making framework for electrical power planning in Ontario

Micro-level decisions relate mainly to the "nuts and bolts" of project assessment and implementation. Decisions at this level deal largely with "backyard" or project-specific issues pertaining to exposure and effect (cause-effect) processes arising from project development. The task here is to characterize risks and uncertainties associated with project development, assess their significance, and apply suitable mitigative or compensatory measures to ameliorate, or at least minimize, any adverse impacts on the host environment/community. While this would seem quite straightforward, there may be considerable information gaps or scientific uncertainty about the environmental impacts of the project. As a result, micro-level decisions are sometimes based on incomplete data and the best judgement of experts.

It is important to point out that the sequence of moving from one level of decision-making to the next may not be as straightforward as suggested in Fig. 1. In essence, there will likely be a "meso" or transition stage where discussions will focus on both broad social issues such as the definition and assessment of energy alternatives, as well as project-specific details.

This chapter examines these two levels of project decision-making from the proponent's viewpoint, and suggests areas where more information is needed and where more research should be directed to deal with inherent uncertainties. More information and more focussed research will help to improve both the reliability and success of future EAs. EA planning and assessment for new or upgraded electrical generation facilities will be used as a case-in-point in these discussions.

ELECTRICAL UTILITY PLANNING - CAUGHT IN THE CROSSFIRE

In a speech to the Canadian Electrical Association's Annual Meeting in Jasper, Milan Nastich (1984), then President of Ontario Hydro, suggested that electrical utilities "are caught in the crossfire of competing social, economic, political and even scientific theory". He went on to note that more than any other industry, electrical utilities (particularly publicly-owned utilities) often find themselves at the heart of conflicts which reflect not only on local or project-specific concerns but also on broader social issues. In many instances, electrical utilities represent an important planning and regional development instrument for government. Electrical utilities, Nastich suggests, are expected to serve every segment of society, and at the same time, they must attempt to conduct their planning and operations to satisfy a myriad of government, customer, labour, environmental and social interest group expectations. The result is that someone or some group feels less than entirely satisfied. To plan wisely, a utility must not only estimate the future demand for electricity and the amount of generation or transmission that will be needed to supply it reliably, but it must also take into account growing financial, regulatory and socio-environmental constraints and uncertainty. Striking an appropriate and acceptable balance is often difficult, and always open to scrutiny.

Electrical utilities are expected to supply electricity to meet the demand; this activity contributes to the day-to-day risks to which all of us are exposed. At the same time, electricity provides numerous benefits which we usually take for granted on a daily basis. In most cases,

however, utility-related risks and benefits are spread unevenly throughout society; that is, some segment of society is asked to bear the risks (i.e. a generating station or transmission line), while all of society benefits. This argument is used frequently by opponents of electrical facility development in Ontario and elsewhere, and is the basis for the NIMBY (Not-In-My-Backyard) syndrome. On a broader level, some would argue that some utility decisions are putting the whole of society at risk for the benefit of no one. This point is often made by anti-nuclear groups who seek to eliminate nuclear power as a viable option in Ontario's energy future.

To legitimize their planning and development decisions, utilities are continually expected to deal with these broad issues as part of their EA studies and documentation for specific development projects. Both the macro or societal tradeoff decisions and the micro or project-specific decisions are important to ensure that a particular development proposal is achieved in an economical, technically competent and socially acceptable fashion. Figure 1 provides an overview of the broad scope and complexity of the EA decision-making process that is employed in planning for new electrical supply facilities in Ontario.

The remainder of this chapter considers in more detail the information needs of a proponent in planning and developing electrical generation facilities at these two levels of decision-making. An objective is to characterize the nature of uncertainty and risk inherent at each level, and to suggest how it can be more effectively assessed and managed.

MACRO-LEVEL PROJECT UNCERTAINTIES

Stemming from the mid-1970s experience with the Royal Commission on Electric Power Planning (RCEPP) and the Select Committee on Ontario Hydro Affairs, both the government and public in Ontario have demonstrated an increasing interest in the broad, front-end societal issues that influence utility planning and development (RCEPP 1980). In general terms, these issues relate to project need (normally associated with expansion of the Bulk Electrical System [BES] in Ontario), as well as the identification and assessment of reasonable project alternatives.

Project need has been traditionally based on long-range economic and electrical load forecasting models. These have a great deal of uncertainty associated with them because numerous other social and political factors complicate these forecasting exercises. Depending on the complexity of the project, and the controversy surrounding it, planning and approvals lead times for major BES facilities in Ontario (i.e. from the time of project inception to approval to build) can range anywhere from 5 to 9 years (Malvern 1984). Because lead times have become so attenuated, new uncertainties often come into play when judging a project's viability. For example, significant changes in the political climate and associated regulatory initiatives can take place within a period as short as five years. Planning timeframes of government agencies are often out of step with long-range electrical system planning activities. Public attitudes may also shift noticeably. The rapid rise of acid rain to a level of pre-eminence since the late 1970s is a classic example of how fast political and social priorities can change. In addition, with long lead times, the potential is increased for technological change

affecting the ways electricity can be produced and distributed. Therefore, the options available at the time of project implementation may differ substantially from those deemed "reasonable" at the time of project planning and assessment.

The definition and evaluation of "reasonable" alternatives at the broad system level has been, and continues to be, the source of particular difficulty for electrical utility proponents. The strategy of the proponent has been to confine his initial considerations to a fairly limited range of technically viable alternatives. In order to meet reliability criteria, the approach has usually been to develop mostly conventional energy technologies, giving some marginal consideration to the introduction of advanced fuel types (e.g. hydrogen, solar, wind). The public, on the other hand, generally adopts a broader perspective, resulting in demands for the proponent to include and evaluate other reasonable alternatives (i.e. soft energy technologies) within its system planning evaluations. From the public's viewpoint, the preference for an alternative often centres on the perceived risks and benefits of a given option.

Often the public's perception of what activities are the most risky differs widely from the scientifically calculated or actual mortality associated with a given activity. For new, unproven technologies, public acceptance has usually been reserved; taking a "guilty until proven innocent" viewpoint. Public acceptance of an idea is a complex process involving more than simply the digestion of information. Emotion also plays an important role in determining the acceptability of risk. Often fears regarding risks are based on single events (e.g. Three Mile Island) and magnified out of proportion to the real risk they pose. Sidall (1981) attempts to explain the mechanisms causing this distorted magnification using what he calls a "public fear loop". One should remember that how we perceive risk tends to be a highly individual trait which is very impervious to objective measure or external judgement (Lerch 1980). Attitudes to risk are based on past experience combined with new information. These attitudes change slowly. A great deal of information about benefits may be needed to counteract a high risk perception. Experience has shown that as new technology becomes more familiar, and is seen to be beneficial within a realistic range of risk, public acceptance generally increases.

Nowhere is the difference between perceived and real risk more obvious than in the case of the electric power industry, particularly its nuclear sector. Although numerous studies have been presented to the public over the past few decades, explaining the safe, low-risk attributes of nuclear power, the industry continues to be high on the public's short list of perceived risks. Public distrust remains steadfast despite studies showing that nuclear power is one of the few technologies that has been subjected to risk analysis and strict regulation throughout its development and implementation; and that emissions from fossil plants (leading to acid rain) as well as risks associated with conservation (e.g. radon emissions in "tight" homes) pose health risk greater than nuclear-related risks (Olds 1984). In the case of nuclear power, there has developed a marked mismatch of perception between the technical expert trying to quantify risks objectively, and the public reacting intuitively to risks. The result has been a lack of public acceptance for the industry which continues to be a focal point in front-end planning discussions for new generating facilities in Ontario and elsewhere.

In order to effectively reduce uncertainty and enhance the public acceptability of various technological alternatives, it is necessary to develop broad-based, scientifically-supported assessments that specify the comparative risks associated with each option. These assessments should routinely characterize and evaluate the risks, benefits and costs associated with various supply/demand options available to the utility planner. They require information from various sectors of the scientific and regulatory communities to enable an evaluation of the complete consequences of each technology to be made. However, this type of cross-sectoral information is often difficult to integrate into a useful format for decision-makers or the public.

Ballard and Hall (1984) identify two basic objectives for these broad-based type studies:

(a) to inform the public and decision-makers about the potential consequences of a decision to develop a particular technology;

(b) to identify, evaluate and compare alternative strategies for dealing with problems and concerns likely to arise when a particular technology is developed.

Decision-makers have been accustomed over the years to judging the acceptability of most projects on a project-by-project basis, using specific yardsticks mainly limited to an expression of anticipated benefits compared to costs (e.g. return on investment). This narrow project perspective tends to mask many of the risks and negative impacts inherent with specific technologies. For example, as Olds (1984) notes, recent attempts in North America to conserve petroleum products have identified the use of smaller, lighter cars and the insulation of homes as very effective conservation measures. However, both activities have several negative impacts which are often ignored. Olds (1984) estimated that the total risk of these activities is significantly greater than other viable alternatives. This reinforces the need for assessments that are broad-based and (for energy supply/demand technologies) extend over the total fuel cycle associated with a particular option (i.e. cradle to grave). The viability of an energy option should be judged not only in terms of quantitative risk/benefit or cost/benefit, but also on the basis of what its function, performances and contributions are as part of the overall energy systems in a social structure. The complexity and dynamics of these types of assessments will dictate that they be regularly updated and evaluated.

MICRO-LEVEL PROJECT UNCERTAINTIES

Once a project's need has been reasonably justified, the proponent's next task is to assess and implement it in a manner consistent with a range of techno-economic, political, social and environmental objectives. The objectives are often expressed and interpreted through a project review under the (Ontario) Environmental Assessment Act, (US) NEPA or a similar statute. At this level of decision-making, assessment and implementation are usually confined to project-specific or "backyard" type details, although it is not unrealistic to expect broader "need-related" questions to arise during site-specific deliberations. In a number of

instances, an attempt has been made to strictly separate the "need" question from project-specific details by using a staged approvals approach (Ontario Hydro 1981a; Shaeffer 1979). Using this approach, a Plan EA is prepared which discusses the need and justification for a project or technology, as well as any relevant alternatives and generic issues. With Plan EA approval in hand, a project EA is then prepared focussing on specific details related to project implementation and mitigation. To date, this approach has met with only limited success, mainly due to procedural difficulties (Wortman 1983).

Effective assessment of a specific project requires an appreciation, recognition and evaluation of impacts likely to occur during particular development activities. It also requires an ability to identify, assess and balance the impacts related to the various feasible and reasonable alternatives (e.g. design, mitigation) to fulfilling the objective of a given project. More often than not, our ability to assess impacts is limited by a great deal of scientific uncertainty related to the processes that will determine the type and magnitude of changes accompanying project implementation. These processes fall into 2 main groups: exposure processes and effect processes.

These processes are usually evaluated on a scientific or expert basis by EA practitioners (left hand side of Fig. 2) and in terms of social acceptability by the public and government (right hand side of Fig. 2). Morgan (1981) notes that the assessment of these processes as they influence a project's acceptability is often fraught with uncertainty. He suggests that uncertainty arises when:

1. The values of the important variables involved are not or cannot be known, and precise projections cannot be made;
2. The physics, chemistry and biology of the processes involved are not fully understood and no one knows how to build adequate predictive models; and
3. The processes involved are inherently probabilistic, or at least so complex that it is infeasible to construct and solve predictive models.

Morgan (1981) goes on to observe that the information regarding these (risk) processes usually falls along a continuum from widespread availability to little or no available information. Unfortunately, given the nature of many of the project-environment interactions (particularly ecosystem concerns), risk information is largely unavailable and usually only of speculative quality. Cairns and Dickson (1980) suggest a few reasons for this. Firstly, many projects are based on technological innovations that are untried as operating systems, and therefore, direct evidence of environmental impact is not available. Secondly, our understanding of the way complex, variable ecosystems function and are structured is primitive, and as a result the potential for error is substantial. Lastly, the development of reasonably sound risk estimates for a large development project requires a multi- and inter-disciplinary effort, particularly between the engineers and biophysical scientists. The record of cooperation and collaboration in this area is not an enviable one. Multi-jurisdictional responsibilities related to large development projects serve to further complicate this situation. In addition to the difficulties identified by Cairns and Dickson, another

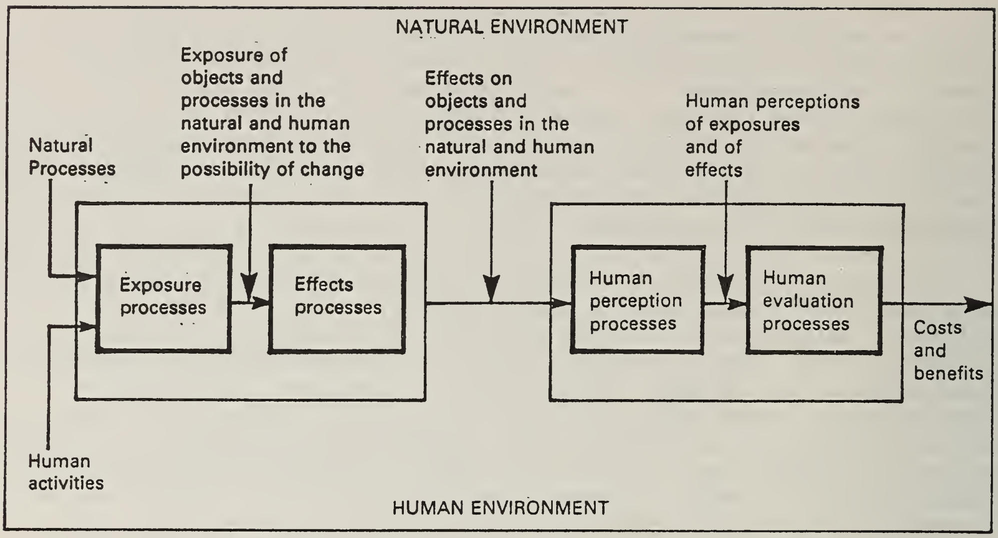

Fig. 2. Environmental exposure and effect processes contributing to risk
Source: Morgan 1981

major problem is that the consequences of an activity on a host environment are frequently not immediate. It may take many years for the severity of a problem to reveal itself. A classic and frequently cited example of this is the mercury contamination problem in the English-Wabigoon River system in northern Ontario. Mobility of human and wildlife populations often complicates the tracking of cause-effect relationships over prolonged manifestation periods.

In addition to uncertainties related to the mechanics of environmental cause-effect relationships, there exists considerable uncertainty as to the irreversibility of the impact that result from project development. It is important to realize that many environmental changes tend to be irreversible by nature. For example, the logging of a wilderness area almost certainly assures that the area's original character cannot be restored, at least over the near-term. Therefore, when addressing long-term environmental impacts, whose bio-physical processes are not well understood or where the role of future technology changes is uncertain, one must consider not only the uncertainty in the economic value of the lost environmental attributes but also the potential irreversibility of these impacts.

Given the reality of these difficulties, it is often necessary to make project decisions on less than adequate information. Decisions made by EA practitioners must often be value judgements, and must trade off the protection of human health or a host environment <u>versus</u> the other valued goods and services that public monetary and other resources can buy (Schrecker 1984). Deciding on the correct balance in this area is not always easy. Indeed many of the adverse impacts we predict in EAs are not borne out by experience - or at least any experiences to date. The EA practitioner must take his scientific assessment and integrate the public's and politician's perception of what is socially palatable. In

most cases, EAs have erred on the side of human safety and environmental protection, and have, as a result, become inherently conservative in their approach. As noted above, this conservatism attempts to deal with a number of uncertainties facing the EA practitioner. It is intended to raise the comfort level of the public with respect to a given project or technology as well as deal with the general lack of "perfect information" relating to the environmental impact(s) of a project.

The EA process deals with these types of uncertainties in a number of ways. Inherent in most recent EAs is some provision for mitigation, compensation or monitoring of anticipated impacts. For most large energy generation projects undertaken by Ontario Hydro since the late 1970s, the utility has negotiated Community Impact Agreements (CIAs) with communities hosting these facilities. These CIAs provide for direct compensation of immediate impacts (e.g. road upgrading, services) and also set out a mechanism to monitor community impacts over the life of the project. Monitoring of environmental changes is also frequently a condition of approval for large projects. Monitoring studies are undertaken to verify a project's compliance and predicted environmental impacts, and to ensure that even low probability impacts are not overlooked. These studies provide an ongoing mechanism for regulators to deal with uncertainties relating to project impacts. However, such studies tend to be long-term and expensive. For instance, it has taken 10 years and millions of dollars to confirm various hypotheses related to the use of once-through cooling (OTC), and to justify the original decision not to use cooling towers on the Canadian shores of the Great Lakes (Ontario Hydro 1981b).

Better follow-up studies, as well as ongoing generic studies at major or similar projects, can also serve to reduce scientific uncertainty over time, and thus increase our level of confidence in predicting project impacts. However, it is also important that proponents and regulatory decision-makers consciously determine the value of this additional information to the decision-makers for a given project. In many instances, the input of additional information may be inconsequential to the decision ultimately made; yet as noted above, to gather these data may take years and be extremely expensive. Also, by the time one has achieved this state of "perfect knowledge", it may be too late to reconcile serious damages. This is particularly true for complex environmental problems such as acid rain.

North and Balson (1984) examined methods of determining the value of information for decision-making related to the question of acid rain control. Their approach develops a series of control/no control scenarios, and attaches probability of impact and cost estimates to them. This method could be developed to have broader application to evaluating the need for additional information related to similar environmental decisions.

As we noted above, information availability is not the only factor that contributes to uncertainty in decision-making. The perceptions and comfort level of the public and politicians are equally important. For many environmental decisions, it is questionable whether additional scientific data would really change what a proponent is forced to do in terms of mitigation/compensation or follow-up monitoring. In many cases, proponents are required to make significant design and other changes to cope with some perceived level of concern and/or to provide an allowance

for unknowns related to his project. This has certainly been the case in the nuclear power industry.

MANAGING UNCERTAINTY IN THE EA PROCESS - FUTURE INFORMATION NEEDS

Effective management of uncertainty/risk during the EA process requires that certain information be readily available to the proponent and the decision-maker. Figure 3 is an attempt to summarize the general types of information/data flows that are needed to reduce uncertainty as one moves through the various levels of project EA decision-making. The following sections discuss some techniques that can help project proponents to reduce uncertainty related to macro and micro-level decision-making.

MACRO-LEVEL DECISIONS

Planning scenarios

As noted earlier, the viability of available technological options for meeting the demand for electricity may vary significantly with time. The acceptability of these options will be dependent on a myriad of related economic, social, political and even environmental factors. The dynamics inherent in these factors result in a great deal of uncertainty in the front-end planning process. In an attempt to characterize and deal with these types of uncertainties, Ontario Hydro has recently developed a new approach to aid in its long-range planning. The approach utilizes various "scenarios" to identify a range of possible conditions that might confront the utility over the next 10 to 20 years (Burke 1983). A scenario is essentially a narrative description of potential alternate future conditions ("what ifs") that may prevail over time. Mechanics of scenario construction and the role of scenario development in corporate planning are outlined in Fig. 4. Scenarios are used to define, test and refine plans, identify opportunities and threats, and to provide a context for decisions and strategy development (Mandel 1983).They are designed to help answer questions about how much electricity will be needed in the future and when it will be required. They bring together diverse kinds of information and views about the future and avoid the pitfalls of looking at the future only in terms of one variable, such as economics. Scenarios also force planners and decision-makers to confront uncertainty by making practical assumptions and considering alternatives. A series of scenarios has been developed dealing with possible economic, technological, political and socio-environmental trends affecting Ontario. Initial attempts at scenario development have centred mainly on four alternative economic futures including: sustained growth; delayed growth; unsustained growth; and stagnation.

Within the context of these broad "economically-driven" scenarios is incorporated information on political, social, environmental and technological trends that is seen to complement the basic logic of each scenario. Effective scenario development requires good information. It is important for these scenarios to be developed by a multi-disciplinary team, using more or less a "brainstorming", interactive process. Sound scenario construction requires good "intelligence" levels relative to political moods and policies, regulatory agendas and priorities, demo-

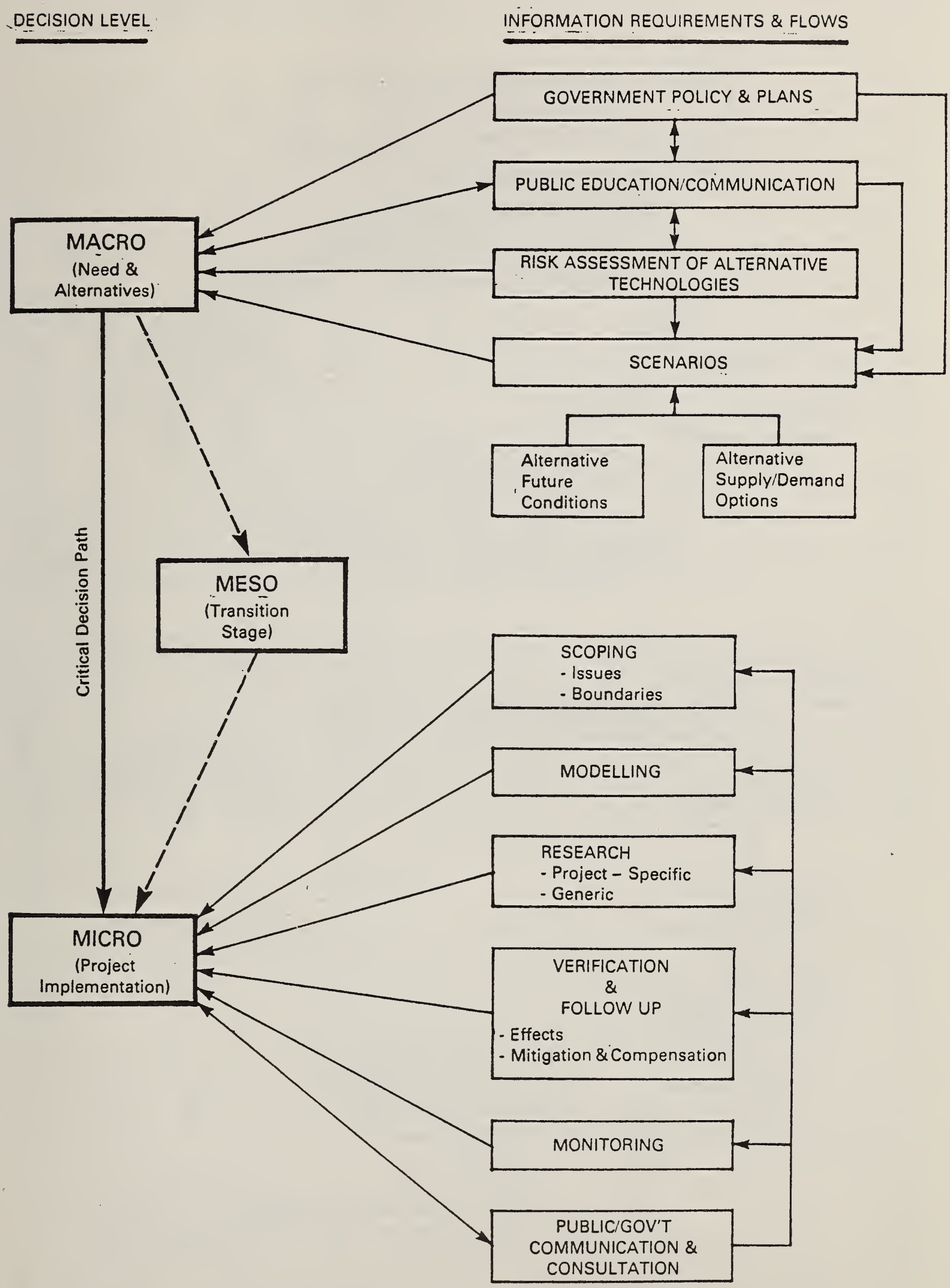

Fig. 3. Information requirements for managing uncertainty in EA decision-making for major electrical utility projects

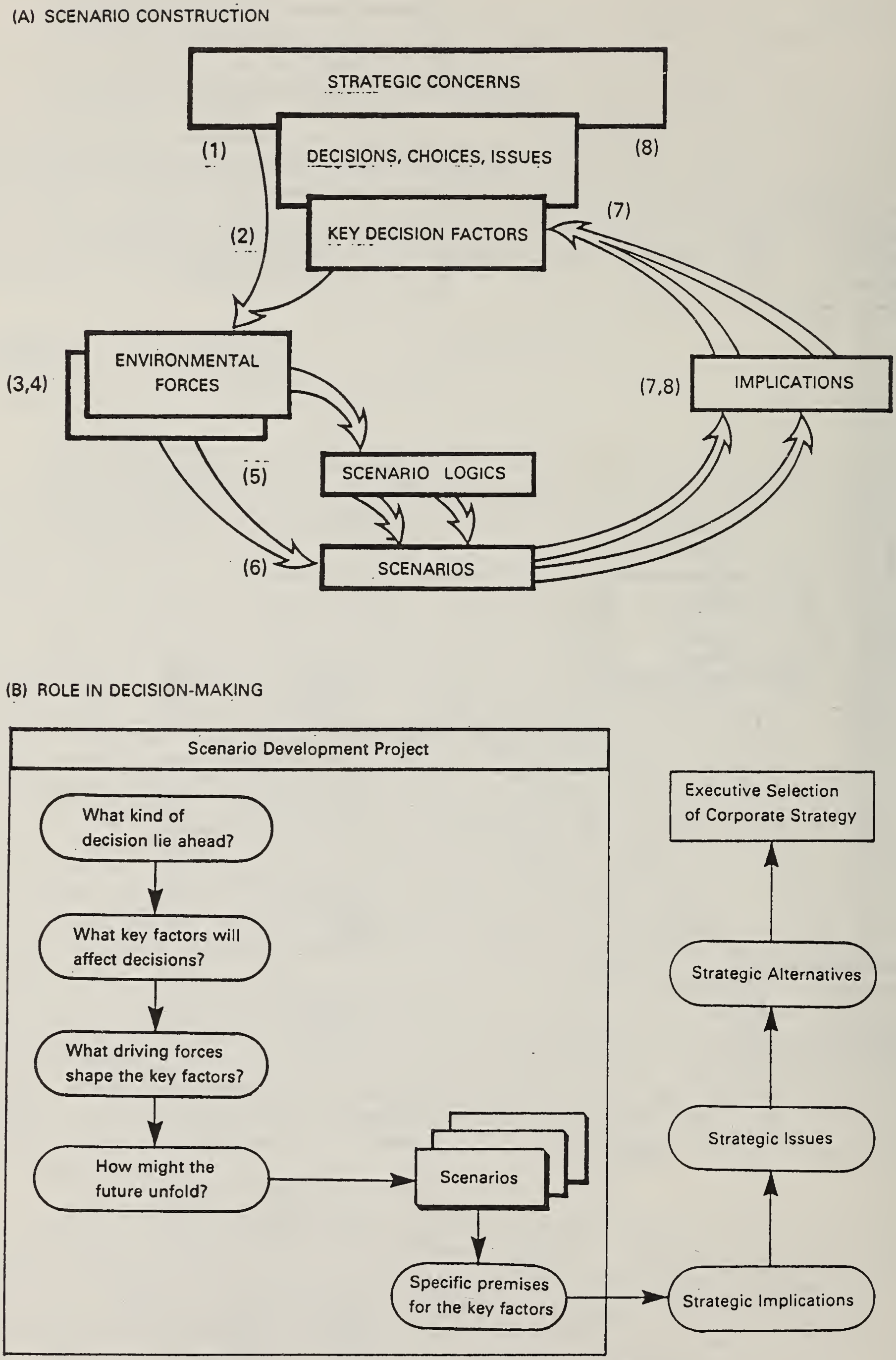

Fig. 4. Scenario construction and role in corporate decision-making
Source: Burke 1983

graphic trends, public interest groups viewpoints, the status of and potential for technological advances, as well as a knowledge of short- and long-term biophysical processes that might influence electrical energy production.

Subsequent to the introduction of the four economically-based scenarios, some work was recently done to develop an "environmentally-driven" scenario. "Environment" in this case is used in its broad sense to include both the physical and man-made environment. The underlying logic of this scenario is a growing dissatisfaction with traditional institutions and decision-making processes; particularly in the areas of consumer protection, resource management, energy use and environmental protection. Environmental and social profiles were developed which complement this basic logic. Specific social, political and environmental trends are then traced through the logic over a twenty year time-frame broken into 5-year time spans. An integral part of this exercise was the identification and assessment of environmental issues that could potentially influence long- and short-term corporate planning and development activities (Paterson 1984). The growth of interest group pressure and the "green" movement are important themes within the overall logic of this scenario. The scenario highlights certain key socio-environmental issues which could influence future planning, particularly in terms of project approval requirements and times, as well as fuel mix considerations. The next step will be to determine the probability associated with these trends, and how best to respond to them using various electrical supply/ demand options in order to minimize long-term uncertainties.

Well-constructed scenarios should help to identify critical impacts and minimize uncertainties related to macro-level utility planning by at least putting realistic bounds (i.e. setting the probable extremes) on future conditions affecting most planning decisions. Long-range planning can then be adjusted to effectively accommodate these alternative futures. Decisions made should, therefore, should be expected to be more resilient and flexible in the light of future uncertainty. These scenarios will require frequent evaluation and scrutiny to ensure that they adequately reflect the most current and realistic visions of the future.

Broad-based conceptual assessments

The use of scenarios helps to reduce the uncertainty related to the question, "how much electricity will be needed?" by considering a broad range of possible future conditions and constraints. The logical next step is to determine "how this demand will be best met?". This will usually involve the type of broad-based conceptual assessments that were discussed above in this chapter. In 1984, Ontario Hydro initiated a broad-level study to examine the full range of supply/demand options available for meeting future electrical needs. For each option, the study will consider its standard costs, environmental impacts, financial impact, macro-economic impact, social acceptability and flexibility (in terms of planning and approvals lead times required). Innovative ways of integrating this type of cross-sectoral input will be required to ensure that all pertinent information is taken into account in the decisions made. These basic data will be utilized to perform an initial screening of all options by deriving a qualitative measure of risk associated with each option. Options surviving this initial screening process will be subjected to a more rigorous analysis that will consider more project-specific details,

including alternative sites. It is intended that this comprehensive and systematic evaluation will form the basis for establishing a defensible long-term system planning strategy for new generation designed to meet future electricity needs. The availability of such a study will be an invaluable reference document for EAs prepared for individual components of such a plan.

MICRO-LEVEL DECISIONS

Scoping

An important step towards recognizing the uncertainties inherent in an EA study is an understanding of the spatial and temporal context within which a project will be undertaken. Figure 5 outlines the basic project-environment interactions ("the ecosystem approach") that should be considered in assessing the impact of a power plant on a given ecosystem. This approach explicitly identifies cause-effect relationships and is fundamental to determining the type of information required to do a risk/impact assessment (Cornaby 1981). Flow diagrams of this type can be expanded to look at a range of possible pathways for extremely fine degrees of decision-making (Couillard 1984).

Construction of a basic ecosystem model not only helps one to appreciate the complexity of the interrelationships involved, but should also help to isolate those pathways along which the most critical impacts are likely to occur. As anyone who has attempted to do an EA for a large project knows, an exhaustive examination of all components and pathways shown in this model would be expensive, time-consuming and, in many cases, inconclusive. Cornaby (1981) suggests that an "indicator" approach should be utilized. An indicator is a measure that reveals critical components and processes in local ecosystems. These usually involve components or processes for which measurement/trends have been developed either locally or via investigations at existing, similar type facilities. They should also be measures that are meaningful to the decision-makers involved. Possible criteria to be used in assessing these critical components have been suggested by a number of investigators (Cairns and Dickson 1980; Waterstone 1983; Ruggles and Sears 1984). These criteria generally relate to the "sensitivity" or "response capability" of a system (natural or man-made) to project-induced changes. As noted earlier, it will also be important to have a feel for the irreversibility of such changes in order to better determine the overall risk to the environment (Viscussi 1985). Irreversibility of impact will equate with some loss of future opportunity to use a resource or particular ecosystem component and this will have a cost associated with it. Application of these criteria when assessing a project represents a round-about, but probably more understandable, way of determining the probability that a project will have an adverse or beneficial impact. An understanding of the inherent ability of a system to cope with changes, as well as some better estimate of the irreversibility of project impacts, will determine the need for and extent of remedial action required to implement a project in an acceptable fashion.

The wise selection of indicators will permit the best use of available data and should significantly increase the confidence levels upon which project decisions are made. It should also provide an "early warning system" whereby the most sensitive indicators will be affected

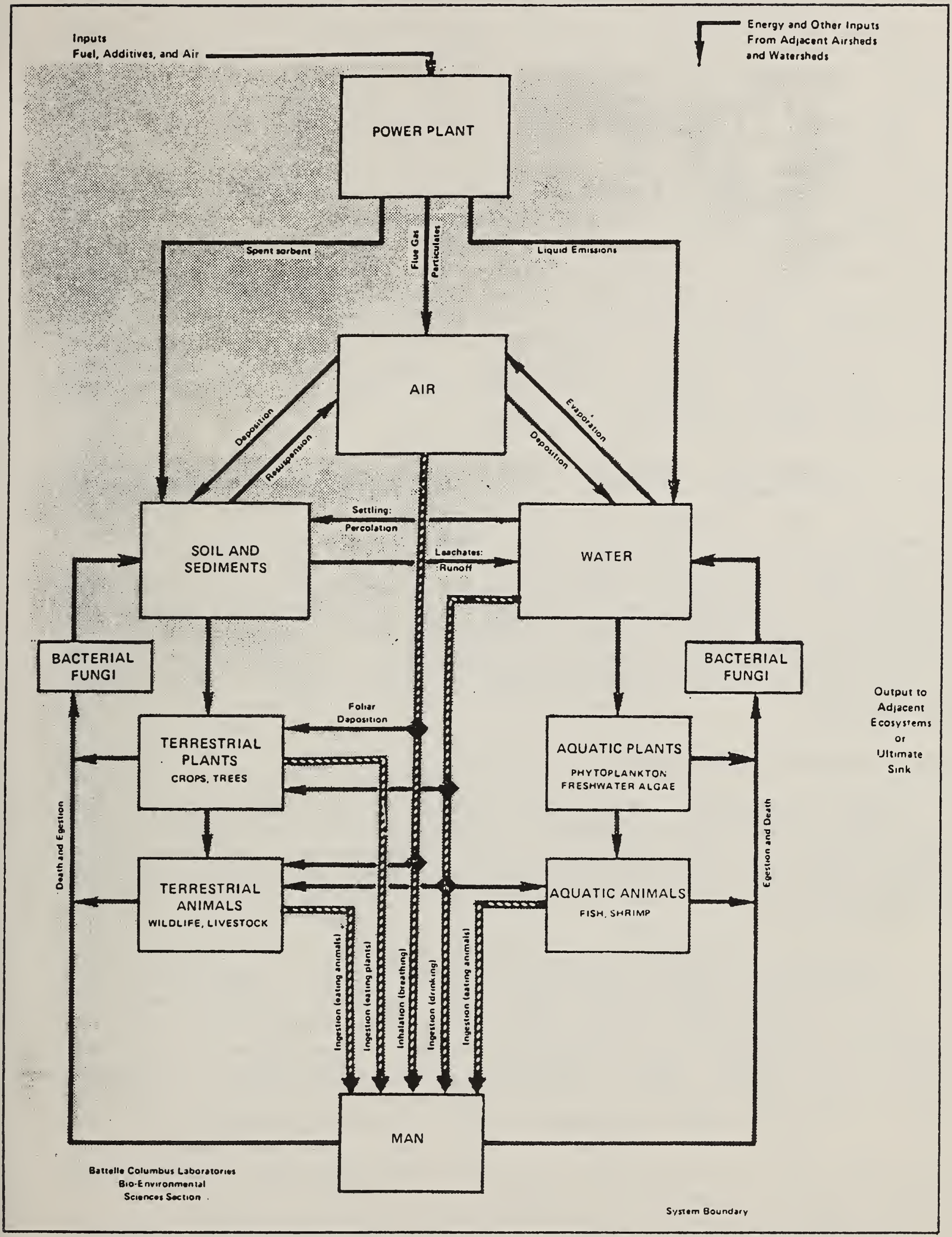

Fig. 5. The components and flows of a basic ecosystem
Source: Cornaby 1981

first, thereby alerting you to potential impacts on less sensitive organisms or processes. By focussing, for the most part, on the key and understandable relationships it should be possible to reduce the level of uncertainty associated with any EAs. This focussing can best be accomplished via some initial "scoping" meetings where scientists, project engineers, government representatives and potentially affected publics get together and discuss the project and its potential implications (Ruggles and Sears 1984). Involving the public usually assures that both "real" and "perceived" consequences are addressed in subsequent EA studies. Studies indicate that the public is generally pre-occupied with the consequences of development activities and not the probability of their occurrence. As a result, high consequence events will likely dominate the public's review and acceptance of any project. This can often be critical to the credibility of this type of study during public hearings. A recent Ontario Hydro pilot project utilized an Adaptive Environmental Assessment and Management (AEAM) workshop as a scoping mechanism. The workshop was useful in conceptualizing and developing a model for the aquatic ecosystem that could be potentially impacted by the cooling system of a planned nuclear generating station. The model was used to establish a pre-operational sampling program (ESSA 1984).

A well-structured scoping exercise is important in determining the nature of information required, and in ensuring that the assessment is carried out in a manageable and cost-effective way. Should substantial information gaps exist about important elements, scoping will allow these deficiencies to be identified, and will enable research/monitoring to focus on these sources of uncertainty, at a time when they can be most beneficial in assisting decision-making. Also, by limiting EA studies and evaluation to key areas of impact and uncertainty, it should be possible to develop more realistic and amenable mitigation or compensation strategies to deal with many of these project-related concerns in a comprehensive fashion.

Impact assessment modelling

Impact assessment involves the prediction and evaluation of the "environmental" consequences expected to accompany the implementation of a particular project. The results of impact assessment are utilized by both proponent and regulatory decision-makers to judge a project's viability and acceptability.

The accuracy of prediction is dependent on an ability to understand how a potentially impacted ecosystem works, and also a familiarity with the type of development being undertaken (and its probable impacts). Information to support predictions is required from various sectors of the scientific and regulatory communities. Direct experience is often not available to support impact assessments. In the absence of tangible evidence related to impacts, EA practitioners have traditionally turned to models to fill in the information gaps. However, inherent in these modelling efforts, is a degree of uncertainty; the confidence limits associated with the predictive capabilities of the model should not be forgotten since a model is not expected to completely duplicate reality under all conditions. Also, most models only deal with a limited part of the affected environment. There is a need, where possible, to integrate these models to take a broader "systems" view of environmental problems.

EA modelling needs to be validated in order to provide confidence in the prediction of impacts. As Shaeffer (1979) notes, environmental assessments that are inherently conservative in their modelling approach may prohibit development due to the prediction of deleterious effects. On the other hand, serious environmental consequences could result if incorrect model predictions allow a potentially disruptive development to proceed. It is critical that uncertainties associated with EA model predictions be explicitly identified, quantified (if possible) and communicated to decision-makers. Often there is more than one model available for application to a particular situation. Shaeffer suggests that the best models for EA purposes are those that make use of available data and have limited requirements for unknown parameter values. An integral part of impact assessment should be a comprehensive and systematic evaluation of models available for EA purposes in order to select techniques that are best suited to the case at hand. Schaeffer (1979) has developed an approach to EA model evaluation. This type of model evaluation has rarely been performed in a comprehensive fashion by EA practitioners or regulatory review bodies. The usual practice in many EAs has been to use models that have gained general acceptance in the regulatory community; and there has been a reluctance to integrate new and innovative techniques. Researchers often become emotionally attached to their own models. The EA community, in general, would benefit from a comprehensive, state-of-art evaluation of EA model availability, applicability and limitations.

An area where EA models can benefit from traditional risk assessment approaches is in determining the probability of certain events that may result from the development of a particular type of project. In most EAs, an extremely conservative approach is taken when formulating impact predictions. Generally, a "worst case" scenario is used; the EA practitioner assumes that a set of routine and upset conditions **will** occur at some point during the development of a project (e.g. oil leaks). He then sets out to model the expected outcomes of these events with little or no attention paid to probability that the events will occur at all. By failing to adequately address probability of occurrence, the proponent leaves himself wide-open to demands for expensive mitigation to cope with events that may have little chance of happening. Decision-makers and the public need to be better informed regarding this aspect of impact assessment; this could reduce their tendency to overreact to the low probability/high consequences (LOPHIC) associated with a project. More judicious application of risk assessment concepts in impact assessment, as well as better recognition of model limitations, is needed to provide more realistic bounds to EA decisions. It should be remembered that given the gaps in scientific understanding, the best that EA has to offer it to draw together and integrate information and opinions from various sectors in order to inform decision-makers about the range and magnitude of possible outcomes of a particular project. It is up to the political and regulatory decision-makers to judge the social acceptability of these anticipated impacts versus the gains of the proponent and society.

EA verification and follow-up

As noted in the previous section, a major limitation to the confidence levels associated with EA studies and decision-making is the lack of in-field verification of EA models and predictions. The value of past experience (i.e. retrospective studies) to provide a basis for more

reliable predictions of impacts has been recognized by a number of EA practitioners and researchers (Hirst 1984; Larkin 1984; Hecky et al. 1984). Follow-up monitoring or auditing of developments to examine, record and analyse actual impacts is seen as a powerful device for aiding in the assessment of similar facilities elsewhere. The term environmental assessment verification (EAV) has been used to describe such studies (Sears 1984). Hirst (1984a; 1984b) notes the increased use of monitoring by regulators and proponents as a means of reducing uncertainties surrounding the approval and implementation of major projects. Newfoundland, for example, has recently amended its EA Act to include a regulation requiring follow-up monitoring on approved projects.

EAV assesses and evaluates the accuracy or effectiveness of hypotheses and predictive/assessment methods used in previous EA studies in order to improve future EAs. It compares predicted versus observed effects from one project, and provides a basis for making adjustments into the assessment process for a subsequent project. It should also examine the adequacy of the scope of enviromental effects that were addressed (i.e. were obvious impacts overlooked?). One important benefit of EAV will be the routine verification of predictive models. EAV should address both the scientific and procedural aspects of the EA process. The delays and uncertainties associated with the approvals process itself (i.e. procedure) should be routinely evaluated and documented to improve the process for future projects. In addition, EAV studies can be carried out to demonstrate compliance with regulatory standards/guidelines; to assess and improve the effectiveness of mitigation/compensation measures in future applications; to improve understanding of the social, political and industrial environment within which projects are developed and, thus, improve understanding of the EA process. Sears (1984) has developed a procedure for integrating EAV into the EA process for major development projects. EAV will significantly expand the data base available for model verification.

Traditionally, funding for these types of studies has been difficult to justify and obtain. Hirst (1984a; 1984b) suggests that widespread support for EAV studies will only develop when it is recognized that uncertainty in the EA process is a factor that has or can have major effects on project decision making or approvals. It must be demonstrated that by better anticipating impacts and mitigation needs via EAV studies, the efficiency of design and operations can be improved through a reduced need to provide "add-ons" to correct environmental problems. In short, the information obtained in EAV studies has value in terms of future project decisions. The systematic verification of EA predictions and methods should significantly improve the confidence levels for future EAs.

Public consultation

A major thread of uncertainty that persists throughout the EA process for a given project is the reaction of the public to the proposal. Different "public" players enter and leave the EA process as it moves from one phase to the next. As noted earlier, there are often significant differences between project proponent and the affected publics, concerning the risks/benefits associated with a project. These differences must be somehow identified, integrated and reconciled in the course of carrying out an EA, in order to effectively manage the uncertainty they bring to bear on the decision-making process. This is often a time-consuming and frustrating task.

Bleiker and Bleiker (1984) have examined the problem of how public perception and polarization affects the success of implementing major projects. From their analysis, they have determined that the prime objective in carrying out EA studies for large controversial projects should be to achieve what they call "informed consent". Informed consent implies that an opponent to your project accepts the basis of your proposal even though it may involve something that will hurt him. Bleiker and Bleiker (1984) have developed a series of 15 steps to aid in achieving this level of support. Adoption of this approach will require an extensive amount of information exchange, in a form that will be meaningful to the groups involved. It will also require some innovation on the part of the proponent to deal comprehensively with the mass of public input which is likely to be received. The job of incorporating public perception information into EA studies for major projects requires increased attention by EA practitioners.

By consciously soliciting imput from the full range of viewpoints concerning his project, the proponent should be in a more advantageous position to judge a project's viability, and plan his strategy to achieve successful approvals and implementation. Hence, the proponent's level of uncertainty sould be substantially reduced (or at least have the bounds better established). Failure to consider the range of public opinion relating to his project, leaves the proponent vulnerable to veto at each decision point in the overall EA process.

CONCLUSIONS

EA practitioners recognize that the EA process is far from an exact science. Unfortunately, those who utilize EA studies for decision-making purposes (i.e. the public and government) often expect the EA process to provide definitive answers. EA is a highly dynamic process - and one that is underpinned by uncertainty and risk. In this chapter, we have attempted to demonstrate how adapting principles of risk assessment offers a more structured medium for characterizing uncertainties in EA decision-making. The level of uncertainty in EA studies can only be reduced by consciously improving the information base of such studies. This will require increased effort to collect and integrate information across sectors of environmental science and policy making. On the other hand, it must be realized that some degree of uncertainty will always persist and it is critical for credibility reasons to document and trace carefully any residual uncertainty throughout the EA process.

EA does not seek to resolve all conflict and uncertainty related to broad societal goals or specific impacts of a project; it can, however, be a powerful organizing instrument for attaining an informed and mutually acceptable compromise about the design and implementation of major projects.

REFERENCES

Ballard, S.C. and T.A. Hall. 1984. Theory and practice of integrated impact assessment: the case of the western energy study. Technical Forecasting and Social Change, 25:37-48.

Bleiker, H. and A. Bleiker. 1984. Winning strategies for the project siting wars. Waste Age, November, pp. 17-20.

Burke, P.J. 1983. Methodology for scenario construction for the scenario development pilot project. Economics Division, Ontario Hydro.

Cairns, J. and K.L. Dickson. 1980. Risk analysis for aquatic ecosystems, in biological evaluation of environmental impacts. U.S. Fish and Wildlife Service, FWS-OBS-80/26.

Canadian Society of Environmental Biologists. 1984. Decision-making - the role of environmental information. Red Deer College, Alberta.

Cornaby, B.W. 1981. Biological pathways, transformation and ecosystems effects associated with power plants, in environmental risk assessment: how will new regulations affect the utility industry. Electric Power Research Institute, EA02964:4-43 to 4-64.

Couillard, D. 1984. New methodology of impact evaluation: coherence graphs. J. Environ. Management, 18:253-265.

Environmental and Social Systems Analysts (ESSA). 1984. Draft report on two workshops to consider the environmental effects and monitoring options for Darlington NGS. Prepared for Ontario Hydro.

Hakanson, L. 1984. Aquatic contamination and ecological risk: an attempt to a conceptual framework. Water Research, 18(9):1107-1118.

Hauptmanns, U. 1984. Risk assessment for interconnected electrical power grids. Risk Analysis, 4(2):97-102.

Hecky, R.E., R.W. Newbury, R.A. Bodaly, K. Patalas and D.M. Rosenberg. 1984. Environmental impact predition and assessment: the southern Indian Lake experience. Can. J. Fish. Aquat. Sc., 41:720-732.

Hirst, S.M. 1984a. Applied ecology and the real world: I. institutional factors and impact assessment. J. Environ. Management, 18:189-202.

Hirst, S.M. 1984b. Applied ecology and the real world: II. resource management and impact assessment. J. Environ. Management, 18:203-213.

Inhaber, H. 1978. Risk of energy production. Atomic Energy Control Board, Ottawa, p. 1119.

Larkin, P.A. 1984. A commentary on environmental impact assessment for large projects affecting lakes and streams. Can. J. Fish. Aquat. Sci., 41:1121-1127.

Lerch, I. 1980. Risk and fear. New Scientist, Jan.3, pp. 8-12.

Malvern, R.J. 1983. The environmental assessment process: application to Canadian electrical utilities, Ontario Hydro. Paper presented to Canadian Electrical Association, Vancouver, March 23.

Malvern, R.J. 1984. Environmental planning and management strategies for Canadian electrical utilities, Ontario Hydro. Paper presented International Seminar on Environmental Sound Development in the Mining and Energy Industries, Chania, Crete, October 21-28.

Mandel, T. 1983. Scenarios: planning under uncertainty. Business Intelligence Program, Stanford Research International.

Morgan, M.G. . 1981. Probing the question of technology-induced risk. IEEE Spectrum, November 1981, pp. 58-64.

Nastich, M. 1984. The electrical utility - microcosm of society. Notes of an address to CEA Annual Meeting, Jasper, Alberta, June 24-26.

North, D.W. and W.E. Balson. 1984. Risk assessment and acid rain policy - a decision framework that includes uncertainty. Presented Conf. Acid Rain: Economic Assessment, December 6, Washington.

Olds, F.C. 1984. Waste heat recovery and conservation. Power Engineering, August 1984, pp. 26-34.

Ontario Hydro. 1981a. Plan-stage evironmental assessment - southwestern Ontario. October 1981.

Ontario Hydro. 1981b. Biological investigations to improve once-through cooling system design for the Great Lakes. Report 81481.

Pate-Cornell, M.E. 1984. Fault trees versus event trees in reliability analysis. Risk Analysis, 4(3):177-186.

Paterson, W.M. 1984. Forecasting of environmental issues affecting corporate activities. Ontario Hydro, Rep. 84006.

Paterson, W.M. and R.G. Ruggles. 1984. Ontario Hydro in northern Ontario - an environmental perspective. Laurentian Review, XVI(2):121-149.

Royal Commission on Electrical Power Planning (RCEPP). 1980. Decision-making, regulation and participation: a framework for electric power planning in Ontario for the 1980's. RCEPP Final Rep., 8.

Ruggles, R.G. and S.K. Sears. 1984. EA scoping requirements. Technical Memo TM84/4, Ontario Hydro.

Ruggles, R.G. 1983. Little Jackfish Hydroelectric project - summary of 1981-1982 definition phase activities and wind-up. Ontario Hydro, Rep. 83107.

Schrecker, T.F. 1984. Political economy of environmental hazards. Law Reform Commission of Canada, Protection of Life Series, Ottawa.

Sears, S.K. 1984. A procedure for follow-up verification of models and methods used to assess and mitigate environmental impact design and development projects. Ontario Hydro Rep. 84084.

Shaeffer, D.L. 1979. A model evaluation methodology applicable to environmental assessment models. Oakridge National Laboratory, Oakridge, ORNL-5507.

Sidall, E. 1981. Risk, fear and public safety. Atomic Energy of Canada Limited, p. 7404.

United States Environmental Protection Agency. 1982. Final programmatic EIS - U.S. Lake Erie Natural Gas Resource Development.

Viscussi, W.K. 1985. Environmental policy choice with uncertain chance of irreversibility. J. Environ. Econ. Management, pp. 28-44.

Waterstone, M. 1983. Toxics and groundwater: the development and application of net risk analysis. The Environmental Professional, 5:46-56.

Wortman, G.L. 1983. Environmental Assessment of provincial/regional power planning schemes. Ontario Hydro, Route and Site Selection Division, Presentation to CEA, Vancouver, March.

CHAPTER 10
REFLECTIONS AND CONCLUSIONS

A.P. Grima and C.D. Fowle
Institute for Environmental Studies, University of Toronto

Since environmental impact assessment (EIA) was introduced in the U.S. in 1969 and in Canada in 1973, it has been widely adopted by other countries, states, provinces and international agencies. The earliest environmental impact statements were narrowly focussed on potential impacts on the biophysical components of the environment but they have been gradually evolving into a more comprehensive decision-oriented process (CEARC 1986). As Malvern and Paterson* point out, the scope of EIA has broadened to include a wide range of concerns including project need and justification, technological and economic feasibility, social and community implications and socio-political acceptability. This evolution is reflected in the current broad definition of EIA from the Canadian Environmental Assessment Research Council (CEARC):

> Environmental impact assessment is a process which attempts to identify and predict the impacts of legislative proposals, policies, programs, projects and operational procedures on the biophysical environment and on human health and well-being. It also interprets and communicates information about those impacts and investigates and proposes means for their management (CEARC 1988).

As a consequence of widened scope, more disciplines are now involved in EIA and there are more stake-holders whose concerns must be addressed if an EIA and the uncertainties associated with it are to be accepted. This broadening of EIA has included consideration of risk estimation and assessment (Beanlands 1986). The Canadian Environmental Assessment Research Council (1986:3) noted that "the determination and evaluation of the risk to human health and welfare from development activities has become a major component of many environmental assessments".

Environmental impact assessment (EIA) and risk assessment (RA) are procedures for predicting future consequences and the likelihood of such consequences. Both are management strategies that attempt to cope with the technical aspects of complex policies and projects; they attempt to provide the best information about risk or hazardous activities, given certain circumstances. Both must also go beyond technical analysis to include social, political and economic considerations.

As Timmerman (Chapter 1) notes, EIA and RA can be mutually supportive and much of the work carried out in EIA already involves risk analysis (see for example, the chapters by Paradine; Dickerson; Duinker; Malvern and Paterson). The more explicit treatment of risk in EIA focusses attention on probability estimates attached to consequences and consequently on the need to arrive at a consensus about the "facts" and the uncertainties attached to them. Therefore:

---

*References to authors, unless followed by year of publication, indicate chapters in this volume.

"... the first beneficial effect of the application of risk to EIA would be to infuse the whole process with the risk philosophy, i.e. outright rejection of the dichotomy of **safe** and **unsafe** with its implied certainties" (Grima et al. 1986:ix).

Generally speaking, uncertainty about both the impacts of the project and the efficacy of mitigation measures is recognized in EIA (cf. Duinker). This recognition of uncertainty significantly extends the scope of EIA; for example, the assessment reflects not only what is known about impacts and mitigation but also the uncertainties (i.e. what is not known). As a consequence, the decision about acceptable risk needs to be based on more than scientific information because there is more scope for discussion about what these "facts" tell us "ought" to be done. Scientific and technical competence would need to be augmented by a process of securing public consensus. The chapters in this volume focus on the importance of giving greater emphasis to risk concepts in EIA. In this concluding chapter we select significant lessons and ideas that may be usefully adopted by proponents, assessors, regulators and public interest groups. We discuss, in turn, some limitations of the application of risk concepts in EIA and some of the ways in which risk assessment may change or extend EIA.

LIMITATIONS

If the general notion of risk assessment has always been part of EIA, what is to be gained from giving it more emphasis? One quick answer is that since uncertainty about impacts and the consequences of technological failure is inherent in all EIAs, risks should be addressed more explicitly and formally. Hence, the methodology specifically designed for risk assessment should be useful in EIA. However, we should be aware of limitations. Risk analysis provides a sharper, more explicit focus but not necessarily more clear-cut results. For example, it is obvious that quantitative, probabilistic calculations of risk can only be done where data are available (see Dooley). The risks described by Paradine for petroleum drilling projects, for example, were only possible because there were data available on the numbers of bore holes and frequency of accidents around the world. Lack of data limited quantitative evaluation of risks to only a few of the hazards associated with petroleum exploration. The quantitative risk estimates included in the environmental impact statements for forest spraying cited by Paradine are limited to potential risks to humans and are based on the findings of toxicological studies with experimental animals extrapolated to humans and on models of exposure in the forest. They are not products of the EIAs themselves. The other area in which probabilistic risk calculations have been used in EA is in the worst case scenarios as described by Dickerson, but even here the calculations are largely based on hypothetical "events".

Generally speaking, there are very few components of EIA which can be subjected to formal, quantitative risk assessments to obtain estimates of probability. This is due partly to the nature of the impacts considered and partly to the lack of data (Paradine; Dickerson). Usually impact forecasts are based on the professional judgement of the analysts and on models which may have varying degrees of verification and, hence, are of varying uncertainty (Duinker 1985). For example, Dickerson notes that

risk assessments are more advanced for health impacts as compared to impacts on the natural environment. Where data are available, one could identify impacts with significant uncertainty and these impacts would be candidates for more detailed risk analysis in preparing the environmental impact statement (Dooley).

Even where it is possible to estimate risks in quantitative terms, it is doubtful that such estimates are the principal concern of decision makers. Paradine suggests that in the case of hydrocarbon exploration, the debates over the probability of a serious spill enhanced the values of the EIAs by encouraging more thorough analysis of the supporting data and that in itself justified the effort. In general, however, EIAs put less emphasis on the probabilities of events than on the type and magnitude of consequences and what can be done to prevent, mitigate or cope with occurrence, i.e. "the conditions that are attached with the approval to proceed" (Duinker). This approach to EIA approvals (or rejection) of development projects also helps to cope with underlying scientific uncertainties.

Consider, for example, the case of the EIA prepared for federal review by Eldorado Nuclear Ltd. in 1978 for its proposed uranium hexafluoride refinery in Blind River, Ontario. The refining process requires an elaborate plant built to high engineering standards. There are many potential environmental and human health risks arising from emissions and accidents. The environmental impact statement (Eldorado Nuclear Ltd. 1978) contains no formal risk analysis of unwanted consequences arising from the plant. Rather, the approach was to rely on professional experience to identify potential hazards and cope with them by means of high standard engineering. Safety devices are designed, in the first instance, to limit scheduled emissions below the levels set by government regulations and secondly, to contain "credible" accidents and process upsets within regulatory limits in the event of an unintended release of chemicals. The possibility of a hazardous release is clearly recognized but estimates of probability are not a concern because the means of coping (mitigation) are designed to contain hazards before they become "risky" as defined in regulations. When considering impacts on air and aquatic systems, scenarios of credible accidents were developed on the basis of professional judgement and the use of models. The extent to which quantitative risk assessments were made by the designers to estimate the probability of failure of construction components is not reported in this EIA but, as Dooley notes, design standards of components take risk of failure into account.

When quantitative data are available, risk analysis adds significantly to the information base for EIA. In the absence of quantitative data, technical treatment of risk could be useful in developing scenarios, models and credible accidents, and sometimes worst case scenarios; and in considering the consequences in each before selecting the appropriate alternative to minimize or prevent occurrence (Dooley). A considered judgement about "probability" is arrived at in the EIA process beginning with expert analysis and passing through the various review stages which follow (Paradine). This process leads to EIA approval (or rejection) after the perceived risks and benefits have been weighed and agreement among decision makers has been reached.

Project proponents and decision-makers need to ask what the consequences are of making the wrong choice. This points to a major practical difficulty in using EIA as a decision tool because proponents need to estimate the degree of risk associated with:

(a) the scientific uncertainty on which the EIA is based; and
(b) the main recommendations (alternatives, options).

Malvern and Paterson make a useful distinction between two main levels of decision making: macro and micro (Fig. 1, Chap. 9). Macro decisions are related to broad policy issues in which major societal trade-offs of risks and benefits are involved as, for instance, in the choice of technologies to be used in energy generation or in arguing the economic need of a project. At the micro level, project specific decisions are of greatest concern to the community hosting the project. Here specific technological risks, equitable distribution of risks and benefits, mitigation and compensation have to be considered (Grima; Pushchack; Timmerman). Because macro decisions are usually made before the terms of reference of an EIA are drawn up, most EIAs are concerned with detailed consideration at the micro level of questions relating to impacts of the project as put forward by the proponent. This is the level at which risk assessment methods are likely to be most useful. However as Malvern and Paterson point out, the basis for the macro decisions need to be spelled out as well.

Risk assessment has provided a useful technical overlay to EIA which has gained in comprehensiveness and rigour (cf. Paradine; Dickerson; Dooley; Malvern and Paterson). In these technical applications further refinements will follow as more information becomes available and frameworks of analysis are improved (Fowle _et al_. 1988). For example, Malvern and Paterson identify various techniques for filling information gaps: scenarios, scoping exercises, impact assessment modelling and environmental assessment verification. However, the explicit application of risk principles to EIA is not limited to technical aspects (e.g. risk estimation); we need to consider the incorporation of risk concepts in EIA as a process. For example, an EIA may identify complexities in the system such that it is difficult to assign probabilities to breakdowns or when we face LOPHIC (low probability - high consequence) risks. In such cases, the analyst and decision maker are faced with a quandary because professional integrity demands that the uncertainties should be recognized and yet, once these uncertainties are recognized, the public confidence in the risk estimates and in the EIA process are shaken (Dooley). Technical analysis now needs to be integrated with social values, conflicts and possible trade-offs. We now turn to these implications of including risk in the scope of EIA.

EXTENSIONS

AN EXTENDED ORGANIZING FRAMEWORK FOR EIA

Descriptions of frameworks of both EIA and RA parallel earlier formulations of the components of rational decision making (see Krewski and Birkwood 1986; McAllister 1980; Whitney and Maclaren 1985; Whyte and Burton 1982; Grima _et al_. 1986). The procedure in each case is divided into a number of "steps", e.g. identify, characterize and scope the

problem; set out constraints and criteria; enumerate alternative solutions and their consequences; and finally, choose an option and monitor the results. Duinker points out, for example, that the steps in EIA of scoping, forecasting, evaluating and follow-up are parallel to the formal steps of risk identification, estimation, evaluation and management. Dooley presents a framework that divides the analysis up into five stages that parallel the physical process from hazard potential through release, exposure and effect on receptors to evaluation. Thus, in a technical sense EIA and RA have a lot in common. Indeed, quantitative risk analyses were introduced into some EIAs in Canada at least 10 years ago (Paradine) and they have been a requirement in the U.S. under NEPA and regulations of the Council on Environmental Quality for many years (Dickerson). Risk analyses have been most usefully applied to EIA when a narrow range of scientific expertise is required (e.g. assessment of oil well blowout data). On the other hand where a wide range of disciplines is required in risk assessment "the record of cooperation and collaboration..... is not an enviable one" (Malvern and Paterson).

In the sense of process or administrative procedures there are some differences. EIA was conceived from the start as a legislated planning process whereas RA (and also rational decision-making) was developed in scholarly journals and later some aspects of RA were adopted as part of the legislative process (U.S. EPA 1984). The social treatment of risk, in particular, has developed a rich literature (see Timmerman Chapter 6). For example, a substantial literature has emerged on risk perception with significant contributions from cognitive and social psychology (e.g. Slovic et al. 1980; Renn 1981; Tversky and Kahneman 1981) and from the hazards research literature (e.g. Kates 1978; Whyte and Burton 1982). Equally significant is the growing literature on the ethical treatment of risk decisions (e.g. NAE 1986).

Thus, while EIA continues to incorporate technical aspects of risk, there is a lot of scope for more emphasis on risk decisions as part of the social and political process. The reason is that EIA is more than an analytical tool; it was conceived and it has developed as an administrative procedure designed to avoid costly mistakes, mitigate unwanted consequences and achieve a sense of fairness to affected groups and individuals. Several authors in this volume (e.g. Pushchak; Timmerman; Grima) examine potential contributions of the social, political, economic and ethical treatment of risk to the EIA process. Progress in this direction may depend in part on whether conflicts over large-scale projects result in substantial delays and high social costs, e.g. the current public hearings on the long-awaited Ontario Waste Management Corporation plant in West Lincoln Township, Ontario; the attempts to close down some nuclear plants in the U.S. (Time 1989).

Strong public opposition to the siting of hazardous facilities draws attention to the need to establish clearly the benefits associated with risky projects. As Pushchak notes "risk has little meaning independent of its benefits". He argues strongly that the benefits of a project be explicitly identified and estimated and also for a broader process that would lead to a more equitable sharing of benefits and risks. The estimation of benefits and their valuation (e.g. the value of lives saved by a new drug or technology) are fraught with difficulties, including the basic question of who values the benefit, the analyst or those affected by risk (Pushchak). Negotiation with affected communities would address this

issue to a large extent as well as the cognate question of the unequal distribution of benefits and risks (Pushchak; Grima; Timmerman).

Failure to address the two questions of risk benefits estimation and distributive justice has resulted in lengthy processess and public disenchantment with decisions that evaluate risk to human health or the environment (e.g. the withdrawal of Alar from the market, public resistance to nuclear power stations in North America, public concern about toxic waste). EIA mandates consideration of the need for the project and of public participation. Unfortunately the public participation process in EIA is constrained by bureaucratic formalities; for example, the EIA process accommodates formal public hearings better than negotiation, mediation and compensation (Grima 1985), although there has been progress in this direction (Bingham 1986).

One way to incorporate the social treatment of risk in EIA is to develop a better theoretical framework that would include negotiation about the distribution of risks and benefits, mitigation and compensation. Grima *et al*. (1986) suggest that comparative evaluations of past experiences are a more direct and cost-effective approach for incorporating the social treatment of risk into EIA. The acceptance of nuclear power could be a case in point. For example, France derives almost 70% of its electricity from nuclear power plants but resistance in other countries has been more marked. Clearly some risks are more acceptable in some countries, cultures and political systems than in others. More specifically, Grima *et al*. (1986) suggest that retrospective studies should preferably be selected from fields where there is considerable experience, and which are likely to emerge as significant in the future (e.g. hydrocarbon development in the Arctic and off-shore; siting of hazardous waste treatment plants).

COPING WITH RISK IN EIA

EIA as a planning process tends to be characterized by conflict, partly because of the uncertainty in the real-world "experiment" that development projects entail, and partly because in the real world there are winners and losers in most development projects. It should be noted, yet again, that the incorporation of risk concepts will not necessarily reduce conflict but the notion that we all take risks provides a common framework for discourse among the various parties.

Malvern and Paterson note that societal trade-offs are implicit in broad (or "macro-level") policy decisions such as job creation *versus* ecological protection. These types of conflicts tend to plague a project throughout the formal EA process which becomes the arena for conflicting stakeholders to argue their philosophical differences as opposed to what Malvern and Paterson call the "nuts and bolts" of project assessment. The opponents of projects use uncertainty as an argument to shore up their interests (Grima). Unfortunately, uncertainty in EIAs generally arises from lack of information (too few observations, no data, etc.) rather than from a lack of scientific understanding. This type of uncertainty will not be overcome by the use of technical risk assessment (which is also data-demanding!). Public interest groups and regulatory agencies often simply argue that projects should not go ahead in the presence of a high level of uncertainty. In addition, what we know about risk perception,

for example, would help us to anticipate public opposition when risks are perceived to be involuntary, to involve large-scale calamity, to involve children or future generations or to benefit small, privileged sectors of society (see Timmerman Chapter 6). Our understanding of risk also points out potential solutions some of which are already incorporated into the EIA process. For example, community impact agreements have been an innovative feature of Ontario Hydro facility planning since the late 1970s (Malvern and Paterson). These provide for compensation, monitoring and even mediation (Grima 1985). In other words they go beyond participatory therapy towards "sharing power and benefits". Pushchak provides other examples of negotiated agreements. He also lists three conditions to enhance public acceptance of risk decisions: access to information, voluntary acceptance and negotiation with the morally relevant community. Grima (this volume) notes that this formal bargaining and compensation "go at least some way towards making the risk less involuntary and uncompensated, therefore, making the risk more acceptable" (cf. Burton et al. 1982).

One recurring theme in this volume is that in EIA and RA, the "facts" themselves may be in dispute due to scientific uncertainty, alternative assumptions and different ways of presenting findings. For example, experts in risk tend to emphasize expected values (i.e. consequence x probability) of catastrophic risks, thereby "discounting their consequences by their rare probabilities" (Keeney and von Winterfeldt 1986:420). Lay persons, on the other hand tend to focus on the worst possibilities. It is clear that a major challenge to practitioners of EIA is to find a feasible way of combining technical risk assessments with the assessment of affected lay persons.

> "One of the great stumbling blocks to the incorporation of risk assessment techniques into EIA is the mathematical nature of much of risk analysis...We believe that the major responsibility for improving technical presentation, i.e. translating the results and implications of analyses, lies with the technical analysts themselves" (Grima et al. 1986:12).

CONCLUSIONS: TOWARDS IMPROVING OUR COLLECTIVE LEARNING CAPABILITY

The fundamental rationale for EIA, risk management and similar planning procedures is to attain improvements in avoiding the negative consequences (i.e. hazards or risks) of proposed major projects such as airports and hazardous waste treatment facilities as well as changes in regulatory procedures such as the registration of new pesticides or drugs. However, we need to go beyond managing well each risky project in an increasingly technological age. We need to improve our individual and collective learning capabilities to cope with new risks. How we increase our learning capabilities should be the highest research priority partly because improving our performance in individual and collective learning would produce the best results in the limited time (and budgets) available, and partly because this approach would ensure that EIA is not just a succession of assessments that are well done but that we put in place an accountable institutional structure that has the capability to quickly learn from mistakes and improve on successes.

This research could benefit from contributions from a wide range of disciplines. However, there are two integrative emphases that show particular promise. The first is to examine the role of science and

scientists in influencing policy and the quality of our collective capability to make ever-improving public decisions. In this respect, we note that there is a biophysical bias in the information used in environmental decision making. Environmental issues such as acidic precipitation and habitat destruction are typically characterized first in biophysical terms. Perhaps this is inevitable because environmental stresses typically manifest themselves in biophysical terms first and, therefore, require the attention of natural scientists. And when the scientific understanding is clear-cut (i.e. based on replicable experimentation with identical results) or not controversial, the decision makers feel secure to make policy decisions that are likely to be politically accepted. On the other hand, when there are "problems of inference" (Lowrance 1976), then the single-minded focus on biophysical science exacerbates the policy making process because conflicting stakeholders use scientific uncertainty as ammunition in the assessment of the social consequences of projects subject to EIA (Grima). In any case, one unfortunate consequence is that we tend to tag on social, economic and historical contributions on to the research agenda set by biophysical scientists rather than to focus on the failures in the institutional processes which result in environmental stresses. This argument emphasizes a conclusion that we noted above, namely, that the major scope for integrating risk concepts lies in EIA as an institutional process rather than in the technical aspects of EIA.

The second emphasis is on learning to identify vulnerabilities resulting from natural hazards or unwanted consequences of projects and technological changes and to enhance resilience in policy responses. Effective policy needs to convert incentives to use hazard-prone technologies into a disincentive by means, for example, of a compensatory fee (cf. Costanza 1987).These compensatory fees would act as "hazard" signposts to decision makers. For example, we could increase our reliance on nuclear power but there are potential risks and costs, and these should be clearly signposted in terms of an appropriate level of third party liability insurance. The same logic could apply to new pesticides.

This emphasis on incentives as improved signposting again focusses attention on flexible institutions and governance rather than "add-on" technical fixes (e.g. liming lakes, stocking fish from hatcheries) that, at best, are palliative solutions and, at worst, put the community on an ever-accelerating treadmill of environmental despoliation and expensive remediation. This should not be interpreted as an argument to reduce the important role that science and scientific understanding play in reaching competent decisions at a time when we seem to face an ever-increasing array of opportunity and risk. It is simply a plea to focus more on the role of institutional failure and the need for institutional reform.

We have presented critiques and reflections from a broad range of disciplinary traditions and perspectives, from engineering and statistics to community choice and ethics. These critiques and reflections are meant to provide directions for a broader set of considerations in EIA and particularly how the more explicit treatment of risk contributes to more competent, more reasonable and more widely acceptable environmental decision-making. We suggest that an increased emphasis on EIA as a collective learning process provides a stronger basis for technical, scientific and economic analyses of the environmental and health risks of developments projects. This perspective on risk management and EIA offers a pragmatic direction for mutually supportive interaction in these two exciting and fast growing fields of knowledge.

REFERENCES

Beanlands G.E. 1986. Risk in EIA: the perspective of the Canadian Federal Environmental Assessment and Review Office. In: Conceptual Trends and Implications for Risk Research, (Eds.) C.T. Miller, P.R. Kleindorfer and R.E. Munn. International Institute for Applied Systems Analysis, Laxenburg, Austria, pp. 53-59.

Bingham, G. 1986. Resolving environmental disputes: a decade of experience. The Conservation Foundation, Washington, D.C.

Burton, I., C.D. Fowle and R.S. McCullough (Eds.) 1982. Living with risk: environmental risk management in Canada. Institute for Environmental Studies, University of Toronto, Toronto, EM-3.

Canadian Environmental Assessment Research Council (CEARC). 1986. Philosophy and themes for research. CEARC, Ottawa.

Canadian Environmental Assessment Research Council (CEARC). 1988. Evaluating environmental impact assessment: an action prospectus. CEARC, Ottawa.

Constanza, R. 1987. Social traps and environmental policy. Bioscience, 37(6):407-412.

Duinker, P.N. 1985. Biological impacts forecasting in environmental assessment. In: Environmental Impact Assessment: The Canadian Experience, (Eds.) J.B.R. Whitney and V.W. Maclaren. Institute for Environmental Studies, University of Toronto, EM-5:115-127.

Eldorado Nuclear Ltd. 1978. Environmental impact statement for a uranium hexafluoride refinery, Blind River. Federal Environmental Assessment Review Office, Ottawa.

Fowle, C.D., A.P. Grima and R.E. Munn (Eds.) 1988. Information needs for risk management. Institute for Environmental Studies, University of Toronto, Toronto, EM-8.

Grima, A.P. 1985. Participatory rites: integrating public involvement in environmental impact assessment. In: Environmental Impact Assessment: The Canadian Experience, (Eds.) J.B.R. Whitney and V.W. Maclaren. Institute for Environmental Studies, University of Toronto, Toronto, EM-5:33-51.

Grima, A.P., P. Timmerman, C.D. Fowle and P.H. Byer. 1986. Risk management and EIA: research needs and opportunities. Canadian Environmental Assessment Research Council, Ottawa, Cat. No. EN 107-3/5-1986E.

Kates, R.W. 1978. Risk assessment of environmental hazard. Scientific Committee on Problems of the Environment, Wiley, New York.

Keeney, R.L. and D. von Winterfeldt. 1986. Improving risk communication. Risk Analysis, 6(4):417-424.

Krewski, D. and P.L. Birkwood. 1986. Structured approaches to risk assessment and risk management. In: Conceptual Trends and Implications for Risk Research, (Eds.) C.T. Miller, P.R. Kleindorfer and R.E. Munn. International Institute for Applied Systems Analysis, Laxenburg, Austria, pp. 149-167.

Lowrance, W.W. 1976. Of acceptable risk: science and the determination of safety. William Kaufmann, Los Altos, California.

McAllister, D.M. 1980. Evaluation in environmental planning. The MIT Press, Cambridge, Massachusetts.

National Academy of Engineering (NAE). 1986. Hazards: technology and fairness. National Academy Press, Washington, D.C.

Renn, O. 1981. Man, technology and risk: a study on intuitive risk assessment and attitudes towards nuclear power. Nuclear Research Centre, Julich, German Federal Republic.

Slovic, P., B. Fischhoff and S. Lichtenstein. 1980. Facts and fear: understanding perceived risk. In: Social Assessment: How Safe is Safe Enough? (Eds.) R. Schwing and W.A. Albers. Plenum, New York.

Time Magazine. 1989. Shutting down Rancho Seco, June 19, p. 43.

Tversky, A. and D. Kahneman. 1981. The framing of decisions and the psychology of choice. Science, 211:235-271.

U.S. Environmental Protection Agency (U.S. EPA). 1984. Risk assessment and management: framework for decision-making. Washington, D.C., EPA600/9-85-002.

Whitney, J.B.R. and V.W. Maclaren. 1985. A framework for the assessment of EIA methodologies. In: Environmental Impact Assessment: The Canadian Experience, (Eds.) J.B.R. Whitney and V.W. Maclaren. Institute for Environmental Studies, University of Toronto, Toronto, EM-5:1-32.

Whyte, A. and I. Burton. 1982. Perception of risk in Canada. In: Living with Risk: Environmental Risk Management in Canada, (Eds.) I. Burton, C.D. Fowle and R.S. McCullough. Institute for Environmental Studies, University of Toronto, Toronto, EM-3:39-69.

APPENDIX

CONTRIBUTORS

Mr. W.D. Dickerson Deputy Director, Office of Federal Activities (A-104),
U.S. Environmental Protection Agency,
401 M. Street S.W., Washington, D.C. 20460

M.Sc. (University of Washington). Mr. Dickerson has been working in the environmental impact prediction and assessment field for over 20 years with experience as a private consultant and as a manager at the Federal level.

Dr. J.E. Dooley Associate Professor, Faculty of Management,
University of Toronto, Toronto, Ontario M5S 1V4

B.A. (Mech. Eng.), Ph.D. (Industrial Eng.). He is a member of the Faculty of Management teaching Operations Management, Statistics and Decision Analysis. His research has focussed on systems analysis for decision-making applied to urban planning, energy systems, environmental management, and, more recently, risk management in developing countries (e.g. Thailand and Indonesia).

Dr. P.N. Duinker Associate Professor and Chair,
Forest Management and Policy, School of Forestry,
Lakehead University, Thunder Bay, Ontario P7B 5E1

B.Sc. Agr. (Guelph), M.E.S. (Dalhousie), Ph.D. (New Brunswick). Research interests include environmental impact assessment, forest decline, wildlife-habitat relations, forest policy development.

Dr. C.D. Fowle Professor Emeritus, Department of Biology,
York University, 4700 Keele Street,
North York, Ontario M3J 1P3

B.A., M.A. (British Columbia), Ph.D. (Toronto). Associate, Institute for Environmental Studies, University of Toronto; Vice-Chairman, Ontario Pesticide Advisory Committee (Ministry of the Environment); Chairman, Canadian National Committee for SCOPE (Scientific Committee on Problems of the Environment) of the International Council of Scientific Unions (ICSU). Interests in applied ecology, especially risk assessment and pesticide problems.

Dr. A.P. Grima Associate Professor, Geography and
Institute for Environmental Studies,
University of Toronto, Toronto, Ontario M5S 1A4

B.A. (Manchester), M.A., Ph.D. (Toronto). Research interests include public consultation, environmental policy evaluation and the role of institutional instruments in waste management. Teaches environmental management in the Department of Geography and Erindale College. Currently working on institutional aspects of risk management, response to climate change and fishery resources allocation.

Mr. R.J. Malvern Section Head,
Environmental Studies and Assessments Department,
Ontario Hydro, 700 University Avenue,
Toronto, Ontario M5G 1X6

M.Eng. (Western Ontario). As Section Head, Environmental Planning and Assessments Section, he is responsible for environmental assessment programs of electrical generating stations, with a special interest in environmental planning, decision-making and risk management.

Dr. R.E. Munn Institute for Environmental Studies,
University of Toronto, Toronto, Ontario M5S 1A4

Fellow of the Royal Society of Canada. B.Sc. (McMaster), Ph.D. (Michigan). Head of Environmental Program and Deputy Director at IIASA 1984-1989. Research interests include air pollution modelling, environmental impact assessment and design of environmental monitoring systems. Editor-in-chief of SCOPE-John Wiley environmental series and a frequent consultant to UNEP, WMO and WHO.

Mr. P.J. Paradine Director, Environment Sector,
Canadian International Development Agency,
Place du Centre, 200 Promenade du Portage,
Hull, P.Q. K1A 0G4

B.Sc. and M.Eng. Previously he was Director of FEARO's Atlantic Region and chaired a number of environmental assessment panels.

Mr. W.M. Paterson Supervisor - Planning,
Environmental Studies and Assessments Department,
Ontario Hydro, 700 University Avenue, 10th Floor,
Toronto, Ontario M5G 1X6

B.Sc. and M.Sc. (Toronto). He is Supervisor - Planning in the Environmental Planning and Assessments Section and has a special interest in environmental planning and issues management.

Dr. R. Pushchak School of Urban and Regional Planning,
Ryerson Polytechnical Institute,
350 Victoria Street, Toronto, Ontario M5B 2K3

M.E.S. (York), Ph.D. (Princeton). His research interests include environmental impact assessment, the siting of hazardous facilities, and the role of compensation strategies in resolving siting disputes involving risk.

Mr. P. Timmerman Institute for Environmental Studies,
University of Toronto, Toronto, Ontario M5S 1A4

B.A. (Toronto), M.A. (London). Research Associate of the International Federation of Institutes for Advanced Studies and IES, Univerity of Toronto. He has been working in the area of environmental theory and practice for the last ten years. His research interests include technical hazards and emergency planning and he was one of the authors of the official report on the Mississauga Train Derailment of 1979. More recently, he has focussed on global environmental hazards, including global climate warming and sustainable development.

Risk perspectives on
environmental impact assessment